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THE APPLICATION OF SPATIAL ANALYTICAL TECHNIQUES TOWARD THE  
CONSERVATION OF LONG-LIVED MARINE VERTEBRATES: THE PELAGIC  
LONGLINE FISHERY AS A CASE STUDY

by

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Dissertation submitted in partial fulfillment  
of the requirements for the degree of Doctor  
of Philosophy in the Department of  
Environment in the Graduate School of  
Duke University

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ABSTRACT

(Ecology)

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2004

## ABSTRACT

One of the largest problems facing marine conservation is how to fish sustainably, and particularly how to reduce fisheries bycatch (the unintended catch of non-target species). I explored spatially explicit tools and approaches for reducing the bycatch of leatherback (*Dermochelys coriacea*) and loggerhead sea turtles (*Caretta caretta*) and blue sharks (*Prionace glauca*) in the U. S. pelagic longline swordfish fishery. I examined the relationship between loggerhead and leatherback sea turtle bycatch, fishing effort and swordfish catches relative to environmental parameters in both the Atlantic and the Pacific to see if there were specific conditions under which turtles were caught. Loggerheads in both basins were correlated with swordfish catches and sea surface temperatures. Leatherbacks were correlated to deep, flat areas in the Atlantic and to areas with little variability in sea surface height in the Pacific. I used two reserve-selection approaches to assess the efficacy of reducing bycatches of blue sharks and leatherbacks. The rank optimization approach selected areas where the fleet has the highest bycatch:catch ratio as candidates for closure. Up to a 75% reduction in leatherback or blue shark bycatch could be achieved with a 20%-25% reduction in swordfish catches. These closed areas are widely dispersed, and it would be impractical to close fishing in so many isolated areas. The spatial optimization approach preferentially chooses adjacent areas of high bycatch and low catch. In this case, the algorithm selected areas near the Grand Banks. I modeled the redistribution of fishing effort that would be displaced if the areas selected by the latter approach

were closed. There was an inverse relationship between the distance traveled between sets and the swordfish catch-per-unit-effort, revealing that longline fishers exhibit area-restricted search. Bycatches of both leatherbacks and blue sharks decreased for each closure scenario given the redistribution of effort. Thus, given the model of effort redistribution, the proposed closures were effective in reducing bycatch of both leatherbacks and blue sharks. These examples illustrate the application of methods that will guide the design of future bycatch-reduction strategies and the placement and design of marine protected areas by taking advantage of recent developments in spatial analysis.

## DEDICATION

To my husband and best friend, Carlos; and the coolest kid on earth, Benjamin.



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## GENERAL INTRODUCTION

Conservation of marine species and ecosystems requires managing human activities, particularly fishing, which is a significant global enterprise, directly employing approximately 200 million people and accounting for 19% of the total human consumption of animal protein (Botsford et al. 1997). Because fisheries can have large-scale ecological impacts, it is important to develop sustainable fisheries that harvest target species responsibly with a minimum of adverse environmental impact.

Currently, one of the largest problems facing marine conservation is how to fish sustainably (Dayton et al. 1995, Hall 1996, Murawski 1996, Crowder and Murawski 1998). The awareness and concern about fisheries bycatch (the unintended catch of non-target species in fishing gear, which for the purposes of this dissertation, does not always result in mortality) has increased recently, making this an emergent issue in fisheries management and marine conservation (Hall 1996, Murawski 1996, Crowder and Murawski 1998). In particular, long-lived species with low reproductive rates, including marine mammals, seabirds, sea turtles, and sharks, appear particularly sensitive to incidental mortality in fisheries (Wooller et al. 1992, Dayton et al. 1995, Crouse 1999).

Understanding the spatial patterns of fishing effort and bycatch, the habitat and environmental factors associated with these distributions, and the scales at which they occur is essential to mitigate fisheries bycatch. One can look at bycatch management in

a hierarchical manner, assuming first that if the bycatch and target species live in different habitats, separated by time or space, then mitigation is straightforward, and can be approached with spatial management techniques (e.g. time-area closures). If, on the other hand, both bycatch and target live in the same habitats, mitigation becomes more complicated, with solutions falling into the more traditional realm of fisheries management regimes, which include setting quotas for target and non-target species, modifying gear and or fishing practices and, in the worst case (at least for fishermen), reducing effort.

Previous population-level estimates of the magnitude of bycatches have often been made for specific gears and fisheries, without explicitly considering the spatial distributions of fishing effort and bycatch distributions (*e.g.*, Perrin et al. 1994). If space is considered at all, it is usually by aggregating effort and bycatch observations into relatively large spatial strata chosen to minimize the variance in the data, without necessarily reflecting stock boundaries, or distinct ocean habitats. A more appropriate approach might be to choose strata based on an understanding of the underlying ocean habitats and species distributions (*e.g.*, Hall and Boyer 1986). Marine organisms are non-randomly distributed -- often clustered in space-- presumably because of the patchy distribution of their resources and habitats. Similarly, fishing effort is spatially clustered at several scales and is often concentrated in areas of high ocean productivity (Hilborn and Walters 1987, Gillis et al. 1993, Lang et al. 1994, Fogarty and Murawski 1998).

Traditional fisheries management has often focused on controlling bycatch by instituting quotas, modifying gear, or reducing effort. However, the popularity of closing, or restricting, marine areas as a tool for managing and protecting marine species is increasing (*e.g.*, Allison et al. 1998, Lauck et al. 1998, Leidy and Moyle 1998, Ruckelshaus and Hays 1998, Russ and Alcala 1996, Dayton et al. 1995). This has led to the suggestion that large expanses of the ocean be set aside for conservation purposes (*e.g.*, Ballantine 1997, Plan Development Team 1990). However, the effectiveness of marine protected areas to protect target and bycaught species is unclear (Crowder et al. 2000), and among other things, depends on the spatio-temporal co-variability of the target and bycatch species (Hall 1996).

A spatially explicit approach to understanding bycatch patterns and the processes that are related to these patterns will guide the development of bycatch-reduction strategies. In addition, this approach will help clarify whether spatially explicit management strategies are even appropriate for highly mobile long-lived vertebrates, and if so, at what scale these should occur given the distribution of bycaught species and human activities. The objective of this dissertation is to assess the utility of spatially explicit techniques to reduce the incidental catch of long-lived vertebrates in marine fisheries by examining three examples within the U.S. Atlantic pelagic longline fishery.

## *Background*

The Atlantic U.S. pelagic longline fishery is widespread, ranging from the Grand Banks to the offshore waters of Brazil and as far offshore as the mid-Atlantic Ridge (e.g., Hoey and Moore 1999). The fishery primarily targets tuna and swordfish, although catches of mahi and pelagic sharks are not uncommon (Hoey and Moore 1999). Data on fishing effort, catches and bycatches for this fishery come from two sources: logbooks and observers. Logbook data are analogous to Captain's Logs, where the captain reports the activities for each trip. One large advantage to this data set is that it includes a large spatial-temporal coverage. The main disadvantage is the potential for bias (e.g., underreporting of bycatch, misidentification of species). Independent, on-board biologists sample the catch and by-catch, referred to here as the observer data set. These data are less subject to bias, but only a small sample (generally 5% or less) of the fleet is sampled, so there are fewer observations and some concerns about how representative these data may be of the fleet's activities.

Pelagic longline trips can be long (month) or short (days), and range from nearshore to offshore. The basic gear configuration includes floats, which are attached to a mainline, which is where the gangions are attached. Each gangion has one hook attached to it. Longlines are typically about 40 miles long. Sets directed at swordfish are fished overnight at shallow depths (70 – 100 ft) and have a high hook to float ratio (~1-3), whereas tuna-directed sets are set deeper (300 – 1200 ft), during daylight hours and have a lower hook to float ratio (~1:10) (Xi et al. 1997, Hoey and Moore 1999).

### *Focal species*

In addition to targeting swordfish and tuna, the longline fishery also catches many other species incidentally, including sea turtles, marine mammals, birds, sharks and billfish (Hoey and Moore 1999, NMFS 2000, 2001b, Baum et al. 2002). In this dissertation, I focused on three species captured incidentally in swordfish sets: leatherback (*Dermochelys coriacea*) and loggerhead (*Caretta caretta*) sea turtles and blue sharks (*Prionace glauca*).

Atlantic swordfish (*Xiphias gladius*) are managed as two stocks, divided by a line at 5 degrees North latitude (2001 SAFE report). Both stocks are vulnerable to exploitation by the U.S. fleet. North Atlantic swordfish are currently classified by the NMFS as overfished, but overfishing is currently not occurring and the stock is thought to be recovering (2003 SAFE report). South Atlantic swordfish do not occur in the U.S. EEZ, thus are not managed under the Magnuson-Stevens Act (2001 SAFE report). The 2001 and 2003 Stock Assessment and Fishery Evaluation for Atlantic Highly Migratory Species (SAFE) reports list them as fully fished, with overfishing probably occurring, but there is a great deal of uncertainty about stock assessments because of inconsistent catch-per-unit-effort (CPUE) information reported by some fleets.

Leatherback sea turtles are listed as endangered in the U.S. Endangered Species Act and as critically endangered by the IUCN. Sixty six percent (n = 158) of the leatherbacks caught in the longline fishery between 1992-1999 were taken in swordfish



sets in the Atlantic (Johnson et al. 1999). The estimated leatherback bycatch by the pelagic longline fleet exceeds allowable take limits in the Atlantic (NMFS 2001).

Loggerheads are listed as threatened in the U.S. Endangered Species Act and as endangered by the IUCN. In the Atlantic, 83% (n = 273) of the loggerheads taken in the longline fishery were caught in swordfish sets between 1992-1999 (Johnson et al. 1999). In the Pacific, 97% (n = 159) loggerheads were caught in swordfish directed sets between 1994-2000 (NMFS 2001a).

The biology of blue sharks is generally understood. They are often discarded because they have little commercial value (NMFS 1999). Of the pelagic sharks estimated discarded dead in the pelagic longline fishery in 1996 and 1997, 73% were blue sharks (NMFS 1999). This translated to approximately 62% and 19% of the total pelagic shark quota for all U. S. fisheries in 1996 and 1997, respectively (NMFS 1999). Baum et al. (2003) estimated that blue shark abundance has declined by 60% since 1986, and Simpfendorfer et al. (2002) estimated that male blue sharks in the western North Atlantic have decreased by 80% since the mid-1980s. NMFS has not evaluated the stock status of pelagic sharks (a management complex of five species which include the blue shark) since 1993 (NMFS 1999). CPUE for pelagic sharks decreased rapidly in the late 1980s but stabilized in 1992. As of 1999, blue shark CPUE showed a generally declining trend, although CPUE was increasing slightly in 1995. In the Grand Banks and Northeast Coastal strata, the numbers of blue sharks caught were

often as large as or more than the number of swordfish caught (Cramer 1996b).

Currently NMFS includes blue sharks with oceanic whitetip (*Carcharhinus longimanus*), porbeagle (*Lamna nasus*), shortfin mako (*Isurus oxyrinchus*) and thresher (*Alopias vulpinus*) sharks. Blue sharks are not endangered, but the present study takes a proactive approach to provide some baseline data and which could be used to reduce by-catches before they do become threatened or endangered.

### *Approach*

I selected the three examples described below for this dissertation because they represent conservation issues involving highly mobile, long-lived pelagic species. In a larger context, these examples illustrate the application of methods that will guide the design of future bycatch reduction strategies as well as the placement and design of marine protected areas by taking advantage of recent developments in spatial analysis and statistics (i.e., geographical information systems (GIS); remotely sensed data; multivariate, spatially explicit statistical techniques). I also used several tools that are commonly used in other systems or for other applications, but not in the marine realm (i.e., different reserve-selection algorithms, models from foraging theory). I did not standardize effort in any way (e.g., Hinton and Nakano 1996; Goodyear 2003) other than limiting the data I used to include only data from the swordfish fishery.

Hall (1996) suggested that bycatch reduction using time-area closures only work if the distribution of species is stable in time and space, which is in turn, dependent on the

ecology of the species. So, in chapter one, I examined the relationships between bycaught loggerhead and leatherback sea turtles, swordfish catches, fishing effort and several environmental variables to gain understanding of the ecology and interaction of the species and their environment. Chapter one illustrates the importance of including spatial information, and specifically the importance of space (the interaction between latitude and longitude) as a variable, in analysis of species' relationships with their environment. In this case, the environment is entirely contained in the fishing grounds, as little information is widely available on turtle distribution. I wanted to know if there was a way to separate the environment where sea turtles are caught in longlines from the environment where the greatest swordfish catches occur, in the hope this might allow for spatial management measures to reduce leatherback and loggerhead bycatch. I applied an existing spatially explicit technique in a way that is new to fisheries managers to sea turtle bycatch in the longline fishery in both the Atlantic and Pacific Oceans. This was, to my knowledge, the first time a two-species, two-basin comparison has been conducted.

Spatially-explicit management strategies are being used with increasing frequency to reduce fisheries bycatch, but there have been few attempts to evaluate the potential costs and benefits of these conservation techniques. In chapter two, I used two reserve-selection approaches to assess the efficacy of reducing bycatches of leatherback sea turtles and blue sharks in the Atlantic swordfish longline fishery. The rank optimization approach selects areas where the fleet has the highest bycatch:catch ratio

as candidates for closure. In contrast, the spatial optimization approach preferentially chooses adjacent areas of high bycatch and low catches. I ran two algorithms (greedy heuristic and simulated annealing) with and without spatial constraints. These methods offer managers objective and simple means to maximize conservation benefits while minimizing costs to a fishery.

Although several studies have led to the implementation of time-area closures, very few of these have actually examined the effects of the redistribution of the fleet once the closures are put in place. Although there have been studies geared to address the issue of fishing behavior and how it relates to the distribution of fishing effort (e.g., Hilborn and Walters 1987; Gillis et al. 1993, 1995; Dorn 1997,1998; Gillis and Peterman 1998) none have used the concepts of area-restricted search to understand how fishers decide where to set their gear within a trip. In Chapter three, I modeled the redistribution of fishing effort conducted by the U.S. Atlantic pelagic longline swordfish fishery given six possible time-area closures designed to reduce the bycatch of leatherback sea turtles and blue sharks by 25%-75%. I tested whether vessels use area-restricted search patterns in deciding where to set their gear, and reallocated effort following an ideal free distribution pattern.

Finally, I synthesized these results to determine which combination of approaches will have the greatest impact on the reduction of leatherbacks and blue sharks in the U. S.

Atlantic pelagic swordfish fishery. I also suggest how these approaches and tools could be useful in other scenarios, and with other parameters.

## CHAPTER ONE

Spatial analysis of sea turtle bycatch in swordfish longline fisheries

## ABSTRACT

Effective wildlife management on large spatial scales requires an understanding of the relationships between species distributions, habitat characteristics and harvesting effort. This is especially true in the oceans. Pelagic longlines take many non-target taxa, including endangered leatherback (*Dermochelys coriacea*) and threatened loggerhead (*Caretta caretta*) sea turtles. To determine whether the distribution of turtle bycatch and longline fishing effort are associated with specific oceanic habitats, I examined the spatio-temporal patterns of fishing effort and sea turtle bycatch in the U.S. North Atlantic (1992-1999) and Hawaiian-based (1994-2000) swordfish fisheries using GIS and multivariate spatial statistics. I examined several environmental variables that serve as proxies for prey distribution in different capacities. In the Atlantic, the number of hooks set was positively related to swordfish catch ( $P=0.001$ ) and significantly linked to depth ( $P=0.03$ ). Loggerhead bycatch was positively correlated to swordfish catch ( $P=0.016$ ) and was higher at particular sea surface temperatures (SST,  $P=0.01$ ). Leatherback bycatch was also positively correlated to swordfish catch ( $P=0.001$ ) and tended to be higher in areas of homogeneous ocean depth ( $P=0.049$ ). As in the Atlantic, fishing effort correlated positively with swordfish catch in the Pacific ( $P=0.001$ ). Loggerhead catches were higher when swordfish catches were higher ( $P=0.001$ ) and related to particular SST ranges ( $P=0.084$ ). The relationship between Pacific leatherback catches and the variability of sea surface height anomalies ( $SSH_{cv}$ ) was insignificant ( $P=0.109$ ) but warrants attention; more leatherbacks were caught in areas of low  $SSH_{cv}$  variability. Because marine turtles do

not concentrate exclusively along restricted bathymetric habitats and turtle bycatch is positively correlated with swordfish catch, my results suggest that traditional "site-specific" management approaches to reduce bycatch in swordfish fisheries may only provide limited protection for these broadly-distributed species.



## INTRODUCTION

Processes such as competition, predation, and dispersal are affected by spatial patterns, such as the distribution of habitat patches, predators, and prey. In turn, ecological processes give rise to spatial patterns through spatially-dependent rates, such as survivorship and fecundity (Turner 1989, Tilman and Kareiva 1997). Although it is widely recognized that ecological patterns and processes are spatially scale-dependent, ecological studies often fail to fully explore the implications of spatial patterns, primarily because these relationships make the analysis intractable (Wiens 1989, Kareiva 1994, Tilman and Kareiva 1997, Mason and Brandt 1999) and because of the underlying assumption that many closed ecological processes are divorced from regional influences and spatial heterogeneity (Ricklefs 1987). These scale-dependent relationships are especially important in marine conservation, where the scales of oceanographic regimes, human impacts, and patterns of habitat use are often greater than those in terrestrial systems, and where the ecosystems and habitat patches are immersed in a fluid and dynamic medium (NRC 1995, Allison et al. 1998, Ruckelshaus and Hays 1998, Hyrenbach et al. 2000).

Managers and scientists alike must consider the species-specific natural history and the dynamic nature of oceanic systems when designing conservation and management plans for far-ranging marine vertebrates (e.g., Carr 1987, Hyrenbach et al. 2000, Polovina et al. 2001). In particular, an understanding of habitat-use patterns is essential to assess the susceptibility of marine species to human impacts. At a large scale, the

distribution of long-lived marine vertebrates often reflects oceanographic domains and their intersection, which in turn influence ocean productivity (e.g., Sund et al. 1981, Ballance et al. 1997, Tynan 1998), whereas at smaller scales, marine-vertebrate distribution likely reflects the immediate distribution of their food resources (e.g., Kenney and Winn 1986, Fiedler and Bernard 1987, Fiedler et al. 1998). In turn, the distribution of prey species is influenced by oceanographic features, both physical and biological. For example, groundfish on Georges Bank exhibit habitat preferences delineated by specific temperature/depth combinations (Murawski and Finn 1988). Many pelagic fishes (e.g., swordfish, tuna, mackerel, and their prey), and invertebrates (e.g., squid) inhabit nutrient-rich waters along temperature fronts, such as the edge of the Gulf Stream, and density fronts, such as river plumes (e.g., Laurs et al. 1984, Olson and Backus 1985, Podestá et al. 1993). There is increasing evidence that upper-trophic pelagic vertebrates concentrate along environmental discontinuities such as thermal fronts and bathymetric gradients because these are often areas of high marine productivity (e.g., Owen 1981, Kenney and Winn 1986, LeFevre 1986, Davis et al. 1998, Fiedler et al. 1998, Polovina et al. 2000).

Fisheries, the taxa they target, and the protected species they threaten, inhabit heterogeneous environments subject to dynamic oceanographic processes, so a spatially explicit approach is required to develop effective bycatch-reduction strategies. The assessment of species distributions and habitats is challenging because many of the environmental variables used to characterize the marine environment are correlated

with one another. As a simple example, usually the farther away from shore one goes, the deeper the water becomes. Thus, ocean depth and distance from shore are seldom independent, rendering classical statistical techniques inappropriate. Several alternative statistical approaches, capable of both dealing with this lack of independence and incorporating the actual location of the observations *per se*, have been developed in recent years (Manly 1991, Legendre and Legendre 1998). The advent of these geo-spatial approaches has transformed the incorporation of space into analyses (e.g., time-series, population dynamic assessments, etc.) from a statistical nuisance into a rapidly-growing field of study. Nevertheless, because many of these techniques are computationally intensive, their use has spread only after the advent of sophisticated computer technology. Additionally, improvements in GIS software and satellite remote sensing now allow researchers to gather, integrate, and analyze a variety of synoptic spatial datasets from large study areas. All these developments have improved the ability of marine scientists to collect, manage, and analyze data on the distributions and oceanographic habitats of far-ranging marine organisms. Within this larger context, the present study illustrates the application of geo-statistical methods to mitigate fisheries bycatch of far-ranging protected species.

My case study centers on the bycatch of leatherback (*Dermochelys coriacea*) and loggerhead (*Caretta caretta*) sea turtles in the U. S. pelagic longline swordfish fisheries operating in the Atlantic and Pacific Oceans. Leatherback and loggerhead sea turtles are listed as endangered and threatened, respectively, under the US Endangered

Species Act. Longlines are the primary gear used to catch tuna and swordfish in the US Atlantic, Gulf of Mexico and Caribbean basin (e.g., Hoey and Moore 1999), as well as in the North Pacific Ocean (e.g., NMFS 2001a). In addition to catching tuna and swordfish, longlines take many non-targeted species, including sea turtles, marine mammals, sharks, billfish, seabirds and undersized swordfish (Hoey and Moore 1999, NMFS 2000, 2001a, 2001b, Baum et al. 2002).

Leatherbacks and loggerheads combined constitute most of the marine turtles taken by U.S. pelagic longlines, accounting for 94 % and 78 % of the turtle bycatch in the Atlantic (1992 – 1999) and the Hawaii (1994 – 2000) fisheries respectively (Johnson et al. 1999, NMFS 2001a). The estimated turtle bycatch by the pelagic longline fleet exceeds allowable take limits in the Atlantic (NMFS 2001b), and led to the closure of the swordfish fishery in the Pacific (NMFS 2001a). Because 90% of the turtle bycatch in both basins occurs in longlines set for swordfish, this fishery is the focus of my paper.

Little has been published about pelagic marine turtle habitats; however, mounting evidence suggests that their distributions are related to hydrographic and bathymetric features. Marine turtles appear to migrate along predictable habitat corridors delineated by specific environmental features (Morreale et al. 1994, 1996, Sakamoto et al. 1997, Polovina et al. 2000). For example, adult Caribbean and Pacific Ocean leatherbacks are believed to move along shelf-slope (200 – 3500 m depth) areas characterized by steep

relief (Morreale et al. 1994), though their movements in oceanic waters have also been ascribed to the location of predictable hydrographic fronts (Morreale et al. 1996). There is no evidence that loggerhead distributions are related to bathymetric habitats. However, satellite-tracked north Pacific loggerheads migrate along narrow corridors delineated by temperature and chlorophyll fronts (Sakamoto et al. 1997, Polovina et al. 2000). Sea turtles in the southeastern U.S. are often concentrated along fronts, convergence zones and sargassum mats (Hoffman and Fritts 1982, Carr 1987, Lutcavage 1996). However, because frontal systems are frequently anchored on bathymetric features (e.g., continental shelves and slopes), it is often difficult to ascertain to what degree “static” or “dynamic” habitat features influence species distributions in pelagic systems (Schneider et al. 1986, Logerwell and Hearn 1996, Springer et al. 1996).

My objectives were to determine whether the distributions of longline fishing effort and turtle bycatch are associated with specific oceanic habitats, and whether these associations are predictable enough to warrant the implementation of spatial management approaches (e.g., time-area closures, marine protected areas) to reduce turtle bycatch in swordfish fisheries. To achieve these objectives, I employ novel geo-statistical techniques designed to incorporate information on these spatial relationships and the correlations between habitat variables into statistical inference.

## METHODS

The National Marine Fisheries Service (NMFS) National Observer Programs provided information on fishing effort and bycatch reported by onboard fisheries observers. I used data from the Atlantic Pelagic Longline Observer Program (1992 - 1999) and from the Hawaii Longline Observer Program (1994 – 2000) in my analyses. These programs monitored an average of 315,423 hooks / year in the Atlantic and 638,048 hooks / year in the Pacific. Observer coverage averaged 4.6 % of the fishing trips between 1994 and 2000 in the Pacific (NMFS 2001a) and 3-5 % of the fishing trips between 1992 and 1999 in the Atlantic (NMFS 2000). Because longline vessels are very flexible in terms of how they deploy their gear (e.g., depth of set, time of day, target species) and how much gear they use on a given set (i.e., consecutive sets in the same area and for the same target species do not necessarily have the same number of hooks), the unit of fishing effort in this chapter is the number of hooks set instead of the number of sets or trips. I did not standardize effort in any way (e.g., Hinton and Nakano 1996; Goodyear 2003) other than limiting the data I used to include only data from the swordfish fishery.

I obtained fisheries (effort, catch, and bycatch) and environmental data for the Atlantic (0° N, 100° W to 55° N, 10° W) and the Pacific (20° S, 130° E to 75° N, 75° W) Oceans, and performed all analyses for each ocean basin separately. First, I cleaned the observer datasets to remove erroneous observations, including records with incomplete set / haul location information and those sets where the length of the longline exceeded

the spatial resolution selected for the habitat analysis (1°, 100 km). Additionally, I discarded longline sets (and their associated catch and bycatch data) with less than 100 hooks that represented fishing effort from bottom / demersal longlines. I used ArcGIS (Arc Geographic Information System; Environmental Systems Research Institute, Inc. 2001) to process and integrate the fishery and environmental datasets. More specifically, I constructed a spatial database of monthly longline sets, catch and bycatch relative to environmental features. When selecting a spatial projection, one must choose whether to preserve one of four features: shape, area, distance or direction. Due to the nature of my analysis and questions, I chose to preserve area. Therefore, I aggregated all of the data into 1 x 1 degree (100 km x 100 km) squares and used a Lambert Equal Area projection to ensure that all grid cells covered the same area, regardless of their location on the Earth's surface.

In my analyses, I focused on variables that relate the performance of the fisheries (e.g., distribution of effort, target catch, and bycatch) to the surrounding marine environment where they operate (e.g., water masses, depth domains, water flow). I used nine variables: total number of hooks set, total number of loggerhead and leatherback turtles caught (not necessarily dead), total number of swordfish caught, median sea surface height anomaly ( $SSH_{med}$ ), mean sea surface temperature (SST), median depth (Depth), coefficient of variation of the SSH anomaly ( $SSH_{cv}$ ; a measure of the rate of change in SSH per grid cell), and depth contour index ( $Depth_{ci}$ ; a measure of the rate of change in depth per grid cell). These environmental variables directly affect, and thus are used

as proxies for, the distribution of prey species. Additionally, I included the influence of location (Space) in my analyses, to account for the spatial structure of the fishery data and the habitat characteristics.

The number of hooks targeting swordfish deployed within a grid cell during a given month provides a metric of total observed fishing effort. I used the number of turtles and swordfish caught instead of catch per unit of effort statistics (e.g., turtles caught per 1000 hooks) because I did not want to mask areas of high turtle catches and high fishing effort. This is a critical consideration given the protected status of these species. However, I consider fishing effort when comparing the number of turtles caught and the number of swordfish caught in each grid cell, because they arise from the same fishing effort.

By using SSH, one can attain some understanding of what the structure of the water column is, both vertically (e.g., convergence / divergence) and horizontally (e.g., geostrophic flow); i.e., processes under the surface of the water. These properties of the flow likely influence the distribution of swordfish and sea turtles by a variety of mechanisms, including the aggregation of buoyant and weakly-swimming prey at fronts and convergence zones (Owen 1981, Olson and Backus 1985, Polovina et al. 2000). SSH data were obtained from the Colorado Center of Astrodynamics Research (CCAR) at the University of Colorado. The satellite altimetry data were combined by CCAR into one SSH grid every 3 days with a spatial resolution of  $0.25^\circ$  latitude / longitude. Because meso-scale eddies and fronts persist in the order of weeks to



months (Owen 1981, Jacobs and Leben 1990), I used one image in the middle of the month to represent the monthly SSH for each grid cell.

SST is an indicator of water mass distributions, known to influence the dispersion of large predatory fishes and marine turtles (Podestá et al. 1993, Polovina et al. 2000, 2004).

Species-specific thermal preferences and thresholds (e.g., some turtle species cold-stun below 10°C, Morreale et al. 1992) and the association of water-mass boundaries with areas of convergence and prey concentration, are likely responsible for the association of marine predators with specific water temperature ranges (Sund et al. 1981, Micol 1997). I obtained filtered Advanced Very High Resolution Radiometer (AVHRR) Reynolds SST data from the National Oceanographic and Atmospheric Administration- Cooperative Institute for Research in Environmental Sciences (NOAA-CIRES) Climate Diagnostics Center (<http://www.cdc.noaa.gov>). These publicly-available data have a spatial resolution of 1 ° x 1 ° latitude / longitude and are averaged into monthly means, after aberrant observations are removed (Reynolds and Smith 1994).

Water depth influences ocean productivity and prey distributions by enhancing mixing (e.g., upwelling) and giving rise to flow gradients (e.g., convergence zones) (Simpson and Tett 1986, Wolanski and Hamner 1988). I obtained bathymetric data from NOAA's National Geophysical Data Center's ETOPO5 5-minute gridded elevation data (NGDC 1988) for the North Atlantic Ocean and the North Pacific Ocean. I aggregated the 5-minute

bathymetric data into a coarser  $1^{\circ} \times 1^{\circ}$  depth coverage, and calculated the median depth for each grid cell.

Because I aggregated the Depth and SSH data into  $1^{\circ}$  squares, I wanted a measure of the variability of the data in addition to considering the average (mean / median) conditions within each grid cell. Thus, I used the  $SSH_{cv}$  (absolute value [standard deviation / mean] \* 100) and calculated the contour index of ocean depth.  $Depth_{ci}$  quantifies the relative change in sea floor depth (topography) within a specified neighborhood (Hui 1985). A small contour index (i.e., 0%) indicates that ocean depth within that grid cell is uniform, and a large value suggests abrupt changes in bathymetry (i.e., 100%). To assemble the  $Depth_{ci}$  layer, I first created two grids of the minimum and maximum depth. For each cell, I quantified the magnitude of the bathymetric gradients using the proportional change in ocean depth:

$$Depth_{ci} = [ (\text{maximum depth} - \text{minimum depth}) / (\text{maximum depth}) ] * 100.$$

I also considered space *per se* in my analyses to account for other spatially- structured processes not included in my analyses (e.g., the clustering of a certain kind of prey, the depth and strength of the thermocline). This is accomplished by incorporating the X and Y coordinates of the center of each grid cell. In addition to considering the spatial autocorrelation of the variables by including location in my analysis, this spatially-explicit approach allows me to assess the explanatory power of the habitat and space

variables directly and indirectly. That is, the influence of each habitat variable can be quantified before and after the unexplained spatial component has been considered.

### *Analyses*

I used the non-parametric, linear regression technique called the Mantel test (Mantel 1967, Legendre and Legendre 1998, Urban et al. 2002) to assess the explanatory power of the variables singly and in combination, for each of the analyses in this study.

Mantel tests have been used to analyse the spatial structure of populations of different plants (e.g., Legendre and Fortin 1989, Leduc et al. 1992, Fortin and Gurevitch 1993) and to assess the impacts of oil rigs on the distribution of bowhead whales (Schick and Urban 2000). A Mantel test is a regression between two (or more, in the case of partial Mantel tests) variables converted to distance (or dissimilarity) matrices. Each matrix emphasizes the variation of the data by considering the pairwise dissimilarities among sample locations (Legendre and Fortin 1989). Much like a multivariate regression, where one plots the residuals of the original model against a new variable to see how much of the variance is explained by adding the new factor into the original model (partial, or Type III, sum of squares), partial Mantel tests assess the correlation between two variables (matrices A and B) controlling for the effect of a third (or a set of) variable (matrix C). This is essentially done by finding the relationships between A and C, and B and C, taking the residuals of each of these relationships and plotting them against each other to get the Mantel coefficient (Legendre and Fortin 1989). Partial correlations are extremely useful in testing for relationships between variables

that might be obscured by the effects of a third (or more). For example, while a simple Mantel test could indicate a significant positive relationship between an environmental variable and the distribution of a species, a partial test could indicate that this result is not due to direct links between the species and the environment, but rather due to the spatial structure of the environment. Partial Mantel coefficients ( $\rho$ ) are standardized, so their magnitude can be interpreted like a standardized regression coefficient (Urban et al. 2002). These coefficients are often very small yet significant, partially due to the huge number of factors that could be related to the distribution of any given species.

All of the tests in this study are one-tailed, allowing for the interpretation of directionality in the relationships, e.g., samples similar in individual species number are similar in environment. The null hypotheses are that the Mantel coefficients are not significantly different from zero (e.g., samples that are close to each other are more similar; or, in my case locations with high numbers of bycaught turtles are characterized by similar environments), thus rejection of the null hypothesis implies the coefficients are significantly greater than zero. However, because Mantel tests compare dissimilarity matrices, the coefficients do not indicate the direction of the relationship between the variables, rather they illustrate the relationship between the distance matrices. In this case, one distance matrix for each variable was created from the “raw” (untransformed) data using Euclidean distance metrics, and then the matrices were appended as fields into one main file (Urban et al. 2002). To determine the direction of the relationship between the variables, I simply plotted scatterplots of the

variables that yielded significant relationships. Although Mantel tests do not require that samples be independent or normally distributed, they do assume that relationships between the variables are linear (Mantel 1967, Urban et al. 2002).

Significance levels (P values) were obtained by randomly permuting the rows and columns of the distance matrices, and repeatedly calculating a new Mantel coefficient (Legendre and Fortin, 1989). I performed 1000 permutations, and estimated the P value as the number of randomized Mantel coefficients greater than the coefficient originally calculated using the actual observed field data, divided by 1000. For instance, if 50 of the 1000 permutations yielded a coefficient larger than the one derived from the observed data, the P value = 0.05 (Manly 1991). For this chapter, I report all coefficients where  $P \leq 0.10$  to account for ecologically important relationships.

## RESULTS

In the Atlantic, the swordfish longline data set consisted of 825,891 hooks set, and 239 loggerheads and 133 leatherback turtles caught. In the Pacific, there were 1,007,172 hooks set, and 138 loggerheads and 30 leatherbacks caught (Table 1.1). In spite of these large data sets, my effective sample sizes decreased once I aggregated the data into 1x1 degree cells. For example, in the Pacific, over one million hooks set were aggregated into 485 grid cells. Overall, the smallest sample size used in these analyses was 28 grid-cell locations for bycatch of Pacific leatherbacks (Table 1.1).

In the Atlantic, the observed swordfish fishing effort was broadly distributed from the Grand Banks, along the U.S. coast, and south to the offshore waters of the Lesser Antilles and Brazil. Loggerhead catches occurred primarily in the Grand Banks and in offshore waters from New Jersey to northern Florida, with additional scattered takes off eastern Cuba and offshore of the Bahamas and the Lesser Antilles. Leatherback catches occurred primarily in the Grand Banks and in isolated clusters along the eastern coast of the U.S., with additional catches off both eastern and western Cuba, and offshore of the Lesser Antilles (Fig. 1). In the Pacific, observed fishing effort was concentrated to the north of Hawaii, extending as far east as 500 km off California. Loggerhead and leatherback catches occurred exclusively north of Hawaii and appeared evenly distributed within the fishing grounds (Fig. 1.2).

To illustrate the information obtained from my spatially-explicit analytical approach, I examine the results for Atlantic loggerheads in detail (Table 1.2). Simple Mantel tests relating the number of loggerheads caught and each fishery / environmental variable (column 1) revealed a significant relationship between the number of loggerheads caught and three variables: the number of hooks set, the number of swordfish caught, and SST. In column 2, I can see that three variables (Depth, Depth<sub>ci</sub>, and SST) were significantly spatially autocorrelated, indicating that at least in the case of SST, the relationship between loggerheads and SST might be spurious, and merely related to the influence of another unexamined spatial component. Column 3 shows that there was a

significant relationship between the number of loggerheads caught and three variables (number of hooks set, number of swordfish caught, and SST), which are not merely caused by spurious spatial effects. However, Table 1.3 (column 1) shows that the number of hooks set is correlated with the number of swordfish caught. In Table 1.2 the results of the partial Mantel tests (Table 1.2, column 4) reveal that the number of loggerhead turtles caught is significantly related to two of the variables (number of swordfish caught and SST), but not to the number of hooks set, as suggested in column 3 (Table 1.2). Mantel test results are often summarized using path diagrams (Legendre and Legendre 1998, Figs. 3 & 4), where the left hand side of the diagram illustrates the relationships between each variable and space (i.e., Table 1.2, column 2), the right hand side shows the partial relationships between the variables (i.e., Table 1.2, column 4), and the arrows reflect the strength of the relationships between all the variables. Path diagrams provide a concise representation of these results, but omit a lot of valuable statistical information which is more amenable to a table format. Thus, for the sake of completeness, I include both path diagrams and summary tables in my results.

In the Atlantic, the number of hooks set was significantly related to the number of swordfish caught within the range of this study ( $\rho = 0.574$ ;  $P = 0.001$ ) and to Depth ( $\rho = 0.029$ ;  $P = 0.03$ ; Table 1.3 column 4, Fig. 1.3a). That is, grid cells with similar numbers of hooks set occurred in (are correlated to) waters of similar depth and where similar numbers of swordfish were caught. Together, these two variables explained 60.3% of the variance in fishing effort ( $0.574 + 0.029 = 0.603$ ). Moreover, scatterplots

suggest that more swordfish were caught when more hooks were set, and that more hooks were set in deeper waters.

Atlantic loggerheads were significantly related to the number of swordfish caught ( $\rho = 0.365$ ;  $P = 0.016$ ) and to SST ( $\rho = 0.206$ ;  $P = 0.010$ ; Table 1.2, Fig. 1.3b). Together, these two variables explained 57.1% of the variance in the turtle catches. Again, this result implies that loggerheads are captured in areas where swordfish are caught, under certain temperature regimes. More loggerheads were taken when more swordfish were caught and, more loggerheads were caught in “cooler” water temperatures (16°- 22°C) than in “warmer” (23° - 32°C) water (Fig. 1.5).

Bycatches of Atlantic leatherbacks were significantly related to the number of swordfish caught ( $\rho = 0.767$ ;  $P = 0.001$ ) and to Depth<sub>ci</sub> ( $\rho = 0.082$ ;  $P = 0.049$ ; Table 1.4, Fig. 1.3c). Together, these two variables explained 85% of the variance in the leatherback catches. These results indicate that more turtles were taken in locations where more swordfish were caught and in areas of flatter seafloor relief (lower ci). Additional scatterplots indicate that these turtles were mostly taken in waters deeper than the continental shelf (depth > 200 m).

In the Pacific, the number of hooks set was significantly related to the number of swordfish caught ( $\rho = 0.775$ ;  $P = 0.001$ ) and insignificant, although it warrants attention, with respect to Depth<sub>ci</sub> ( $\rho = 0.032$ ;  $P = 0.118$ ; Table 1.5, Fig. 1.4a). Together,



these two variables explained 80.7% of the variance in the fishing effort distributions. Perhaps not surprisingly, scatterplots showed that more swordfish were caught when more hooks were set, and that more hooks were set in homogeneously deep waters with low  $\text{Depth}_{\text{ci}}$  values.

Bycatches of Pacific loggerheads were significantly related to the number of swordfish caught ( $\rho = 0.370$ ;  $P = 0.001$ ) and to SST ( $\rho = 0.100$ ;  $P = 0.084$ ; Table 1.6, Fig. 1.4b). Together, these two variables explained 47% of the variance in the turtle bycatch. As in the Atlantic, more loggerheads were taken when more swordfish were caught, and cells with high loggerhead bycatch were characterized by “cooler” water temperatures ( $16^{\circ} - 24^{\circ} \text{ C}$ ), instead of warm ( $24^{\circ} - 32^{\circ} \text{ C}$ ) water (Fig. 1.5).

The relationship between the number of Pacific leatherbacks caught and  $\text{SSH}_{\text{cv}}$  was insignificant ( $P = 0.109$ ; Table 1.7, Fig. 1.4c). Although this variable only explained 10.9 % of the variance in the bycatch of this species, it warrants attention as it may be indicative of a real trend if there had been more data. More leatherbacks were caught in areas where  $\text{SSH}_{\text{cv}}$  was low, (i.e., grid cells where there were low SSH gradients), suggesting the turtles were caught in areas with little variability in sea level height, possibly indicative of large (1x1 degree) convergence or divergence zones. To discriminate between these two alternatives, I examined how leatherbacks were distributed with respect to  $\text{SSH}_{\text{med}}$  and found that the turtles were mostly caught in areas where the median SSH anomaly was close to zero. This result suggests that

leatherback turtles were taken from areas where there was little meso-scale variability (e.g., eddies, current meanders).

## DISCUSSION

The spatially-explicit analyses described above have identified true correlations between sea turtle bycatch and environmental factors, thus providing insight into means to mitigate this bycatch. The U.S. pelagic swordfish fishery in both the Atlantic and Pacific Oceans occurs in areas that coincide with major current systems and thermally dynamic meso-scale areas (Podestá et al. 1993, Polovina et al. 2000, Seki et al. 2002). In the Atlantic this fishery is also associated with bathymetric features (i.e., shelf break, Witzell 1999). Using a powerful statistical technique, I was able to tease apart the importance of these spatially autocorrelated variables and the influence of spatially structured fishing effort.

In the Atlantic, fishing effort (the number of hooks set) was strongly related to the number of swordfish caught, with higher swordfish catches in areas where more hooks were set. In addition, fishing effort was related to median depth, with more hooks set in areas of shallow waters. I found similar results in the Pacific: the number of hooks set was positively related to the number of swordfish caught. Additionally, fishing effort in the Pacific was negatively related to  $\text{Depth}_{\text{ci}}$ , with 92% of the hooks set in areas of flat relief ( $\text{C.I.} < 50$ ).

Loggerhead bycatch in both the Atlantic and the Pacific was positively related to the number of swordfish caught: more loggerheads were caught in areas where more swordfish were caught. Loggerhead bycatch in both basins was also negatively related to SST, although the temperature range in which most loggerheads were caught was slightly more defined in the Pacific (Fig. 1.5).

In this study, loggerheads were caught primarily between 16° - 22 °C in the Atlantic and between 16° - 24 °C in the Pacific. Polovina et al. (2000, in press) found that Pacific loggerheads spent considerable amounts of time in SSTs of 15° - 25 °C, usually in association with strong geostrophic currents along current meanders and eddies. These localities are often characterized by strong flow gradients and hydrographic fronts, which can be readily identified by specific SST isotherms. Because SST and flow gradients (SSH) are spatially correlated, their influence on the distributions of marine organisms is difficult to tease apart. Using spatially explicit techniques, I was able to quantify the significance of these two variables in conjunction and in isolation, once the spatial structure of the underlying habitat was taken into account. These analyses revealed that SSH was not related to the distribution of bycaught loggerheads, while SST explained as much of the bycatch distributions in this species as swordfish catch.

One possible explanation for this disparity is that turtles cue on SST to find the convergence zones that aggregate their buoyant prey. Changes in surface properties

across fronts are often accompanied by changes in water clarity (Olson and Podestá 1987), which may provide additional cues for locating these areas of prey aggregation. The turtles tagged by Polovina et al. (2000) traveled along two frontal zones associated with the 17 ° C and 20 ° C surface isotherms. Aerial surveys in the Atlantic Ocean recorded 255 loggerheads distributed from the shore to the onshore edge of the Gulf Stream (Hoffman and Fritts 1982). This is an area where several small hydrographic fronts, associated with surface temperatures up to 21 ° C in the winter and 30 ° C in the summer, have been identified (Pietrafesa 1989, Lee et al. 1991). These frontal features are not associated with strong SSH anomalies (e.g., meso-scale eddies), but are related to geographically stationary fronts, likely associated with bathymetric features (e.g., mid-shelf front, slope front), and are characterized by temperatures within the range of SST where fishery observers documented turtle bycatch in both the Hawaiian and Atlantic pelagic longline fisheries.

Bycatches of Atlantic leatherbacks were significantly related to the number of swordfish caught (more leatherback takes occurred in areas with higher swordfish catches) and less strongly to Depth<sub>ci</sub> (more leatherbacks were taken in areas of flat bathymetry). Most (63%) of the leatherbacks were caught in areas of deep water with homogeneous seafloor (C.I. < 40). Morreale et al. (1994) found that satellite-tracked leatherbacks in the Pacific (n = 2) and in the Caribbean (n = 4) followed specific bathymetric contours (ranging from 200-3500 m), and remained close to the continental slope. In the Pacific, eight nesting satellite-tagged adult female

leatherbacks ranged from Costa Rica to the Galapagos (Morreale et al. 1996), across a deep-water region of flat bathymetry. These differences might be ascribed to the disparate temporal scale of the observations. In Morreale et al. (1994) the satellite transmitters had only been deployed for 24 days, whereas in Morreale et al. (1996) the tags operated for up to 87 days. Ten months of satellite-tracking data reveal that an additional nine satellite-tracked nesting leatherbacks were documented to travel from Mexico to Chile across deep, flat expanses of the Pacific Ocean (Eckert and Sarti 1997). Thus while leatherbacks may occupy shelf-shore areas while nesting or transiting through coastal areas, they also cross deep-water areas during their vast migrations.

The relationships between Pacific leatherbacks and the selected environmental variables were less clear, likely due to the small sample sizes used in the analyses ( $n = 28$ ). Though this relationship proved to be insignificant ( $p = 0.109$ ), leatherbacks were positively related to the coefficient of variation in SSH, with more turtles caught in areas with more variable SSH conditions. This relationship might be indicative of a real trend in the data, thus warrants attention.

Swordfish and turtles (except Pacific leatherbacks) are caught by longline fisheries in the same places in both the Atlantic and the Pacific (Figs 1 and 2). Using logbook data from the U.S. Atlantic longline fishery, R. A. Myers (personal communication, May 2003) also found that swordfish catches were the best predictor of leatherback bycatch

in the NMFS Northeast Distant (NED) Statistical Region, an area encompassing the Grand Banks and other fishing locations. In the Atlantic, turtle and swordfish catches are both high in the area off the Grand Banks, along the eastern seaboard from New Jersey to Georgia, and in the waters between Cuba and Haiti (Fig. 1.1). These are areas of high productivity, where the ocean floor rises quickly creating an area of upwelling caused by the “collision” of the deeper currents with the continental slope. In the Pacific, swordfish and loggerhead catches both follow the location of the North Pacific Subtropical Frontal Zone (STFZ), also an area of increased prey aggregation caused by the convergence of different water masses (Seki et al. 2002, Polovina et al. 2004). Prey species of both swordfish (e.g., squid and small fishes) and turtles (e.g., gelatinous zooplankton and squid) are densely concentrated along these regions of enhanced productivity (Wilk et al 1988, Pearcy 1991, Polovina et al. 2004). Thus it is not surprising that these species co-occur in these areas.

The positive relationship between loggerhead catches and both SST and swordfish catches in both oceanic basins led me to further examine the relationship between swordfish catches and SST. Swordfish and loggerhead catches in the Atlantic do not peak at the same SST range (Fig. 1.5a). Eighty four percent of the loggerheads were caught between 16 -22 ° C, whereas only 36% of the swordfish catch occurred in this temperature range. Most (64%) of the swordfish were caught in waters with SSTs between 24 - 32° C (Fig 1.5a). Podestá et al. (1993) found that sets targeting swordfish in the northeast Atlantic occurred within a wide SST range, but were concentrated at

SSTs between 19 -21° C. These results suggest that a decrease in turtle bycatch is possible, at least for the Atlantic fleet, if fishers were to target swordfish in warmer waters (i.e., SST > 22 ° C). Further examination of the data shows that of the loggerheads caught in waters with SST < 22 ° C, all but four were taken in the Grand Banks, an area which is currently closed to longline fishing activities (Federal Register 2002). However, since some turtles are also caught within this higher temperature range (Fig. 1.5), and because we lack data on the fine-scale distribution of sea turtles in pelagic waters, the effects of displacing all fishing effort to these warmer waters (i.e., off the US coast) could be worse for the turtle populations, particularly if higher overall fishing effort were to be deployed in waters >22 ° C to maintain the current swordfish catch levels. In the Atlantic, reduced turtle bycatch could be achieved through a combination of setting longlines deeper in the water column (e.g., Polovina et al. 2003) and in warmer waters. Simulation models under various scenarios (constant catch, constant effort, etc.) could be used to show the effects of the redistribution of the fleet to waters with higher temperatures, but are beyond the scope of this paper.

Unlike the situation in the Atlantic, swordfish catches in the Pacific coincided with the same SST range in which loggerhead bycatches were concentrated (Fig. 1.5b). Eighty nine percent of the loggerheads and 79% of the swordfish were caught in waters between 16 - 22 ° C, and all of the loggerheads were caught in waters between 16 - 24 ° C (Fig 5b). From January-May, the Hawaii-based pelagic longline fleet targets

swordfish in the STFZ, an area characterized by two main fronts associated with the 17 ° C and 20 ° C isotherms (Seki et al. 2002). Bigelow et al. (1999) documented high swordfish CPUE at temperatures ranging between 15 - 20 ° C and in dynamic areas of high frontal energy.

My results suggest that spatial protection measures such as time-area closures are not likely to have a great effect in reducing turtle bycatch by pelagic longline fisheries because these bycatches are significantly correlated with catches of the target species within the extent of the fishing grounds. It may be particularly difficult to target swordfish without taking loggerheads because both the target species and the turtles share the same oceanographic habitats, at least within the extent of the fishing grounds. In the Atlantic, it might be possible to reduce loggerhead bycatch by setting the gear in waters with SSTs greater than 23 ° C, a mitigation measure that would require all of the vessels to be equipped with vessel monitoring systems (VMS). Unfortunately, there does not seem to be a comparable solution for Pacific loggerheads, because the temperature ranges for turtle and swordfish catches overlap completely. In addition, fishing at higher SSTs in the Atlantic would not eliminate bycatches of endangered leatherbacks. Although the number of leatherbacks caught at higher water temperatures was lower than those at cooler waters (Fig 5a), and not all bycaught turtles die, the highly endangered leatherback turtle may not be able to sustain even these bycatch levels (Spotila et al. 2000, NMFS 2000).



My analyses show that Mantel and partial Mantel tests can help discriminate between the effects of location (i.e., space) and the influence of habitat variables (i.e., SST, SSH, depth). For example, fishing effort in the Atlantic was clustered (Fig. 1.1a and Table 1.3), as were swordfish catches (Fig 1b), Depth, Depth<sub>ci</sub>, SST and SSH<sub>med</sub> (Figs 1.3 and 1.4). The spatial autocorrelation of these variables impacts the spatial structuring, or clustering, of fishery catch and bycatch distributions. In addition, these results have important management implications. For example, if I had accepted the significant relationship between the number of hooks set and the number of loggerheads caught (column 1, Tables 1.2 & 1.6) without examining the issue in more detail, the implications for management would have been simple: reduce fishing effort to mitigate turtle bycatch. But the relationship between the number of hooks and the number of turtles caught was not significant once the confounding effects of space and additional habitat variables were considered (column 4, Tables 1.2 & 1.6). Instead, I found a significant relationship between turtle bycatch, swordfish catch, and SST. This result implies that moving fishing effort into warmer waters could reduce turtle bycatch, at least in the Atlantic, if effort is not displaced to an area with higher turtle abundance. While both options will reduce turtle bycatch, they have very different implications for the fishery. The first would reduce fishing effort overall, with consequent economic hardship, whereas the second option suggests that management of this fishery is possible via operational changes. Within the hierarchy of possible solutions ranging from simple (i.e., bycatch and target species are separated in time or space) to complex (i.e., bycatch and targets overlap in time and space), my results

suggest that simply using spatial protection measures such as time-area closures are not likely to have a great effect in reducing turtle bycatch by pelagic longline fisheries because, at least within the fishing grounds, incidental sea turtle takes are significantly correlated with catches of the target species and both the target species and the bycaught turtles share the same oceanographic habitats. The interdependency of the species needs to be taken into consideration, and a messier solution is warranted.

To reduce turtle bycatch, fisheries managers will have to focus on operational aspects of the fishery (e.g., depth of the sets) or modifications to fishing gear (e.g., hook type). Based on the dive profiles of two tagged loggerheads in the Pacific, Polovina et al. (2003) suggested that the elimination of shallow (< 100 m deep) sets should reduce loggerhead bycatch substantially. Due to new regulations on the Hawaii-based longline fishery, there was no loggerhead bycatch in deep-set (> 100 m) swordfish longlines from April- December 2001 (Polovina et al. 2003). In addition, preliminary results from a NMFS-conducted experiment in the north-east distant region (NED; a NMFS statistical zone including the Grand Banks), indicate that using 18-0 circle hooks and mackerel for bait may reduce sea turtle bycatch (John Watson et al. 2003).

In this paper I focused on the habitat characteristics of bycaught turtles: i.e., defining those habitats within the region where the fishery operates and where bycatch occurs. I believe that this powerful, novel, spatially-explicit approach will be extremely useful in other studies of marine wildlife-habitat associations because it eliminates spurious relationships between variables that are both interrelated and spatially autocorrelated.

TABLE 1.1. Sample sizes used in these analyses, derived from the NMFS observer datasets.

	ATLANTIC (1992- 1999)		PACIFIC (1994 – 2000)	
	Sample size	No. locations (1° cells)	Sample size	No. locations (1° cells)
Hooks set	825,891	556	1,007,172	485
Loggerheads	239	60	138	79
Leatherbacks	133	65	30	28

TABLE 1.2. Summary of Mantel coefficients (and P values) for multivariate analysis of the relationships between loggerhead sea turtle bycatch, “environmental” dissimilarity and geographic dissimilarity for the U.S. Atlantic pelagic longline swordfish fishery.

	r Loggerhead ~ Env	r Space ~ Env	$\rho$ Loggerhead ~ Env  Space	$\rho$ Loggerhead ~ Env  All
Hooks	0.230 (0.007)		0.230 (0.007)	
Swordfish	0.450 (0.023)		0.450 (0.017)	0.365 (0.016)
Depth		0.523 (0.001)		
Depth <sub>ci</sub>		0.360 (0.001)		
SST	0.171 (0.006)	0.755 (0.001)	0.265 (0.003)	0.206 (0.01)
SSH <sub>med</sub>				
SSH <sub>cv</sub>				
Space		--	--	

*Notes:* r = simple Mantel coefficient;  $\rho$  = partial Mantel coefficient; Env = “environmental” variable(s); Hooks = No. hooks set; Swordfish = No. swordfish caught; Depth = median bathymetry; Depth<sub>ci</sub> = depth contour index; SST = mean sea surface temperature; SSH<sub>med</sub> = median sea surface height; SSH<sub>cv</sub> = median sea surface coefficient of variation; Space = location; | = “given” the inclusion of all of the other factor(s) in the model; notation, *e.g.*, Hooks~Env|All = relationship between the

number of hooks set and one environmental variable given all other variables. Tests of significance are one-tailed. Only coefficients with  $P \leq 0.10$  are included, otherwise they are not listed because they are not significant (n.s.).

TABLE 1.3. Summary of Mantel coefficients (and P values) for multivariate analysis of the relationships between fishing effort, “environmental” dissimilarity and geographic dissimilarity for the U.S. Atlantic pelagic longline swordfish fishery. Table headings same as Table 1.2.

	r Hooks ~ Env	r Space ~ Env	$\rho$ Hooks ~ Env  Space	$\rho$ Hooks ~ Env  All
Swordfish	0.584 (0.001)	0.112 (0.001)	0.579 (0.001)	0.574 (0.001)
Depth	0.052 (0.001)	0.464 (0.001)		0.029 (0.03)
Depth <sub>ci</sub>		0.240 (0.001)		
SST	0.126 (0.001)	0.417 (0.001)	0.089 (0.002)	
SSH <sub>med</sub>		0.059 (0.021)		
SSH <sub>cv</sub>				
Space	0.109 (0.001)	--	--	

TABLE 1.4. Summary of Mantel coefficients (and P values) for multivariate analysis of the relationships between leatherback sea turtle bycatch, “environmental” dissimilarity and geographic dissimilarity for the U.S. Atlantic pelagic longline swordfish fishery. Table headings same as Table 1.2.

	$r$ Leatherback ~ Env	$r$ Space ~ Env	$\rho$ Leatherback ~ Env   Space	$\rho$ Leatherback ~ Env   All
Hooks	0.343 (0.002)		0.343 (0.002)	
Swordfish	0.786 (0.001)		0.786 (0.001)	0.767 (0.001)
Depth		0.528 (0.001)		
Depth <sub>ci</sub>		0.391 (0.001)		0.082 (0.049)
SST		0.655 (0.001)		
SSH <sub>med</sub>				
SSH <sub>cv</sub>				
Space		--	--	

TABLE 1.5. Summary of Mantel coefficients (and P values) for multivariate analysis of the relationships between fishing effort, “environmental” dissimilarity and geographic dissimilarity for the U.S. Pacific pelagic longline swordfish fishery. Table headings same as Table 1.2.

	r Hooks ~ Env	r Space ~ Env	$\rho$ Hooks ~ Env  Space	$\rho$ Hooks ~ Env  All
Swordfish	0.776 (0.001)		0.776 (0.001)	0.775 (0.001)
Depth		0.146 (0.001)		
Depth <sub>ci</sub>		0.112 (0.001)		0.032 (0.118)
SST		0.316 (0.001)		
SSH <sub>med</sub>				
SSH <sub>cv</sub>				
Space		--	--	



TABLE 1.6. Summary of Mantel coefficients (and P values) for multivariate analysis of the relationships between loggerhead sea turtle bycatch, “environmental” dissimilarity and geographic dissimilarity for the U.S. Pacific pelagic longline swordfish fishery. Table headings same as Table 1.2.

	r Loggerhead ~ Env	r Space ~ Env	$\rho$ Loggerhead ~ Env   Space	$\rho$ Loggerhead ~ Env   All
Hooks	0.274 (0.006)		0.278 (0.005)	
Swordfish	0.439 (0.001)		0.444 (0.001)	0.370 (0.001)
Depth		0.353 (0.001)		
Depth <sub>ci</sub>		0.088 (0.073)		
SST		0.231 (0.001)		0.100 (0.084)
SSH <sub>med</sub>				
SSH <sub>cv</sub>				
Space		--	--	

TABLE 1.7. Summary of Mantel coefficients (and P values) for multivariate analysis of the relationships between leatherback sea turtle bycatch, “environmental” dissimilarity and geographic dissimilarity for the U.S. Pacific pelagic longline swordfish fishery. Table headings same as Table 1.2.

	r Leatherback ~ Env	r Space ~ Env	$\rho$ Leatherback ~ Env   Space	$\rho$ Leatherback ~ Env   All
Hooks				
Swordfish				
Depth		0.192 (0.025)		
Depth <sub>ci</sub>		0.405 (0.003)		
SST		0.256 (0.017)		
SSH <sub>med</sub>				
SSH <sub>cv</sub>	0.009 (0.101)			0.026 (0.109)
Space		--	--	

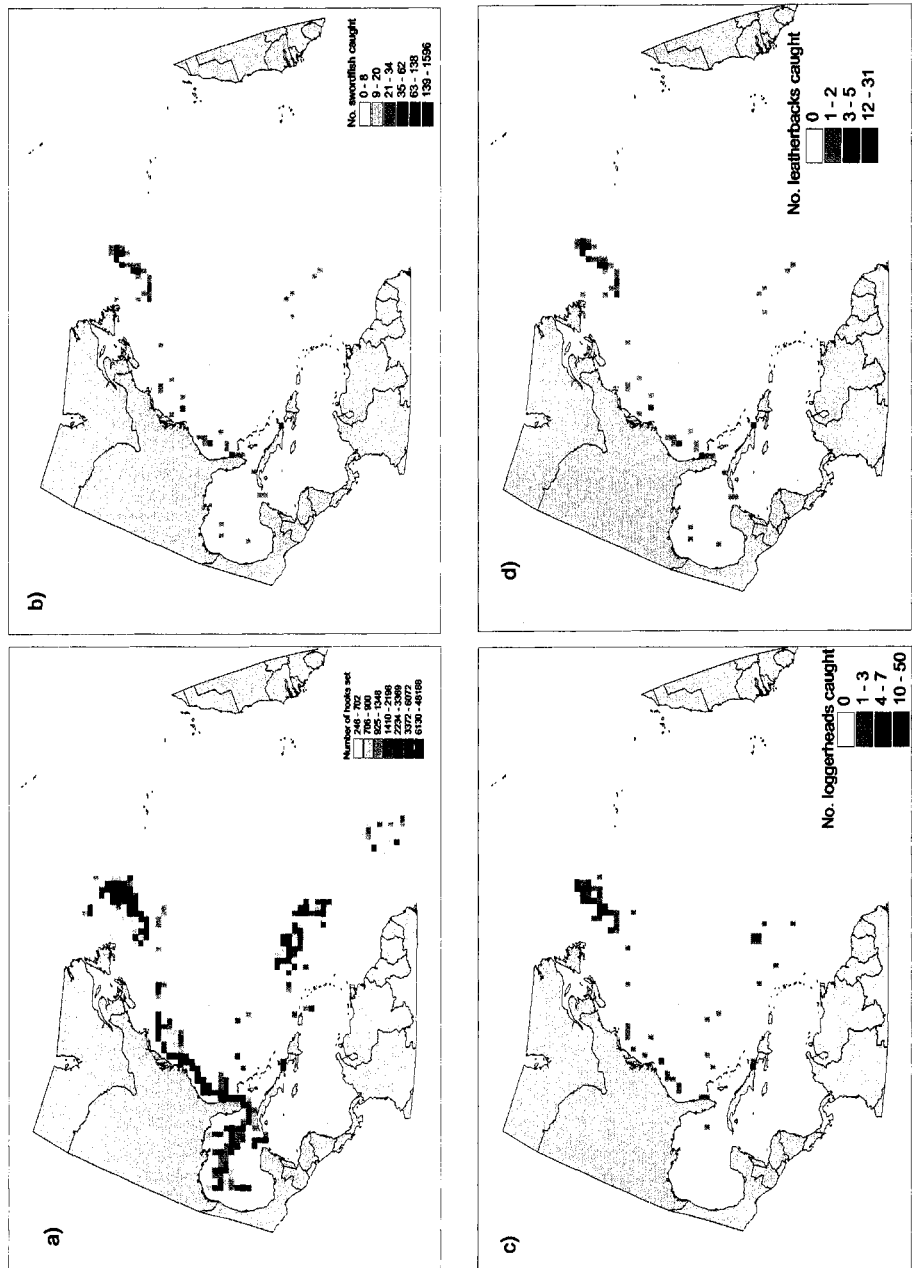


Figure 1.1. Location of observed (a) longline sets, (b) swordfish catches, (c) loggerhead sea turtle catches and (d) leatherback sea turtle catches in the Atlantic pelagic longline swordfish fishery, 1992-1999.

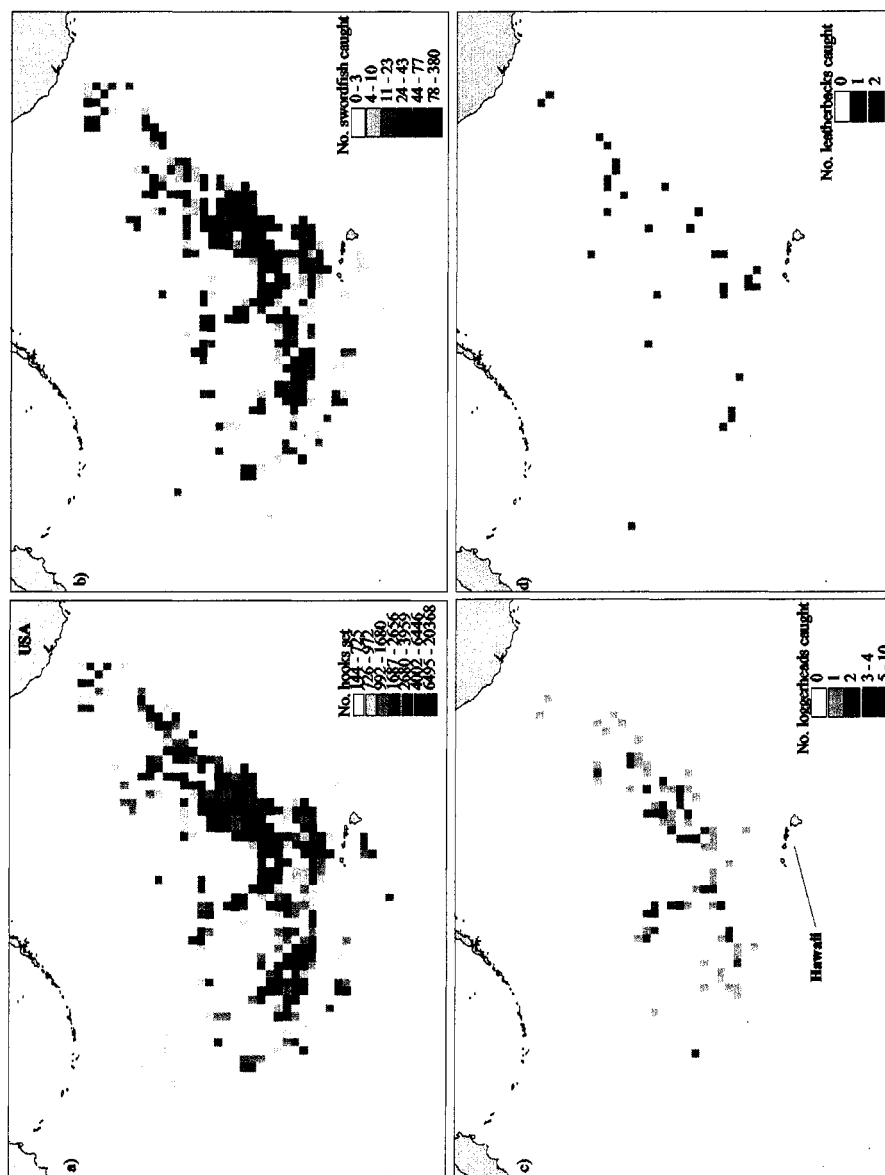


Figure 1.2 Location of observed (a) longline sets, (b) swordfish catches, (c) loggerhead sea turtle catches and (d) leatherback sea turtle catches in the Hawaii pelagic longline swordfish fishery, 1994-2000

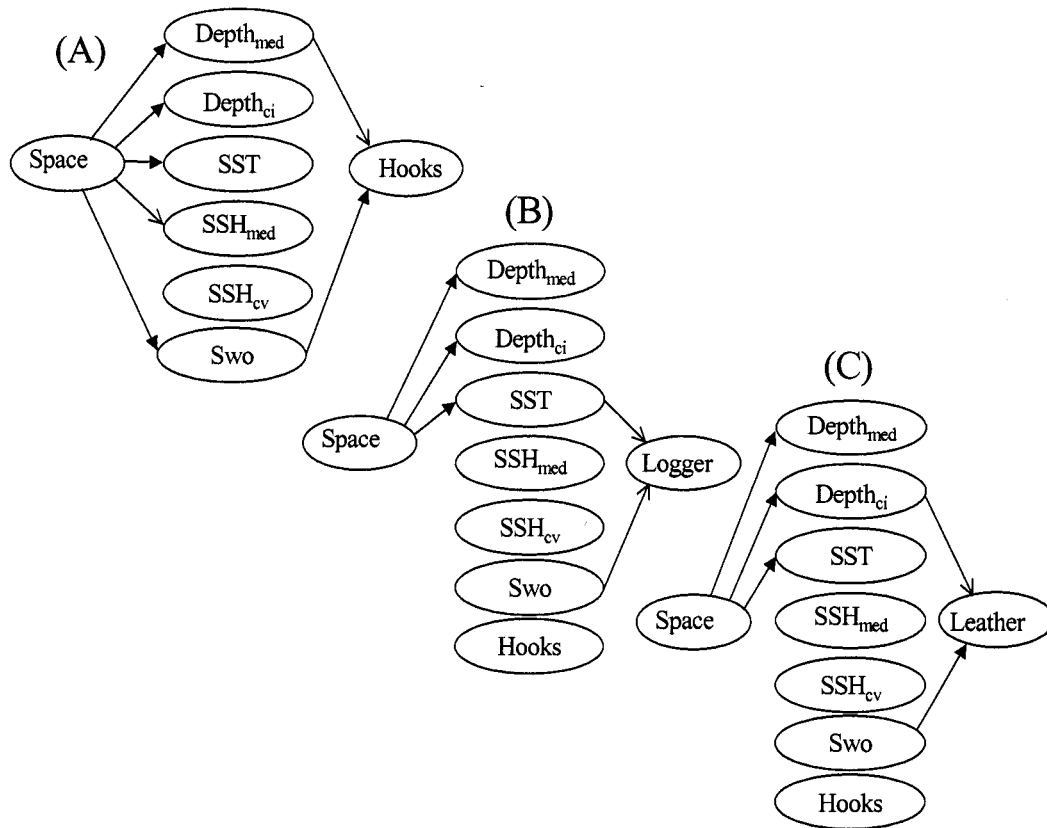


Figure 1.3. Path diagrams of Mantel and partial Mantel tests summarizing the key relationships between (a) longline sets, (b) loggerhead sea turtle catches and (c) leatherback sea turtle catches in the Atlantic pelagic longline swordfish fishery, 1992-1999. Solid line with solid arrowhead =  $0.001 \leq P \leq 0.005$ ; solid line with open arrowhead =  $0.006 \leq P \leq 0.015$ ; dashed line with solid arrowhead =  $0.016 \leq P \leq 0.055$ ; dashed line with open arrowhead =  $0.056 \leq P \leq 0.097$ .

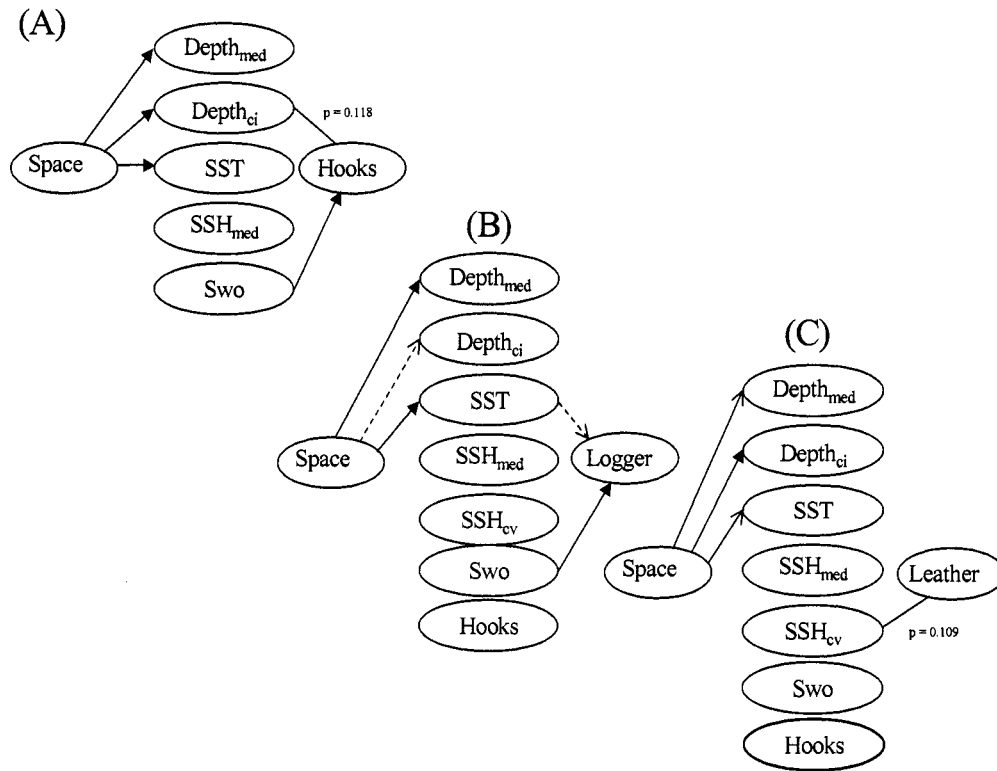


Figure 1.4. Path diagrams of Mantel and partial Mantel tests summarizing the key relationships between (a) longline sets, (b) loggerhead sea turtle catches and (c) leatherback sea turtle catches in the Hawaii pelagic longline swordfish fishery, 1994-2000. Solid line with solid arrowhead =  $0.001 \leq P \leq 0.005$ ; solid line with open arrowhead =  $0.006 \leq P \leq 0.015$ ; dashed line with solid arrowhead =  $0.016 \leq P \leq 0.055$ ; dashed line with open arrowhead =  $0.056 \leq P \leq 0.097$ .

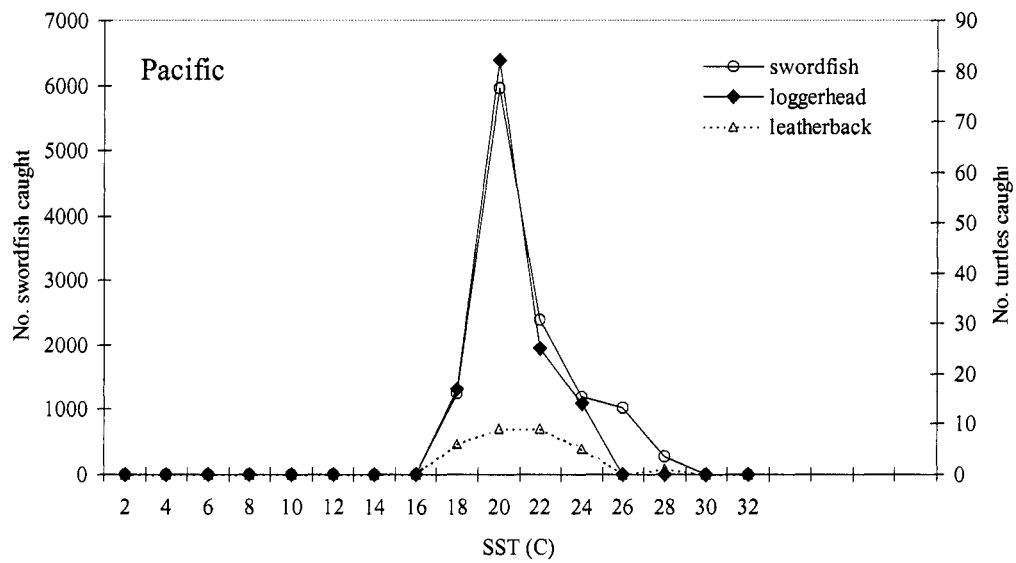
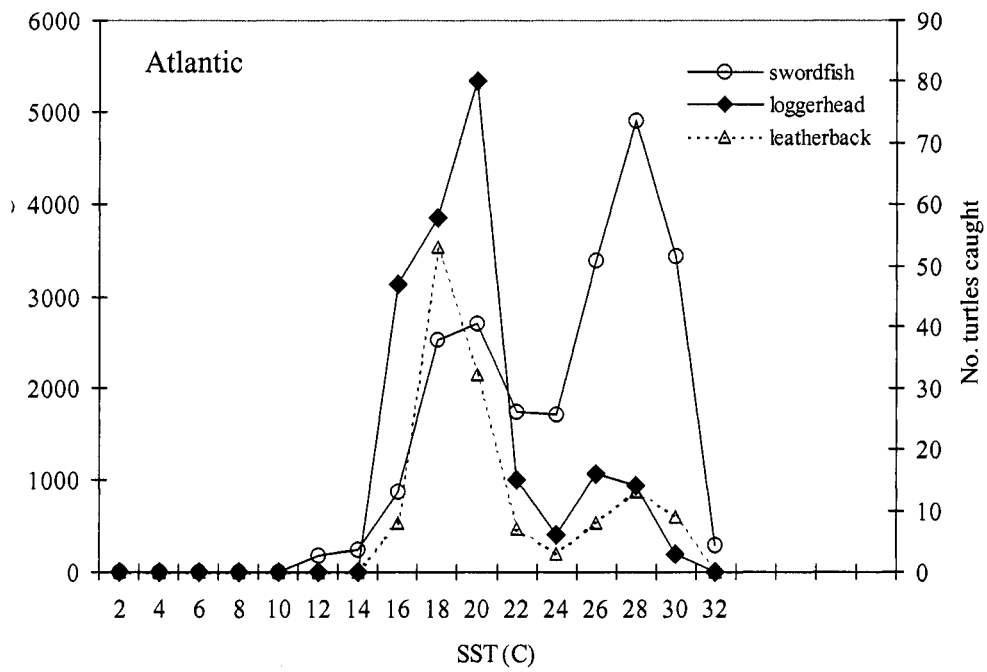


Figure 1.5 Relationship between number of swordfish (*Xiphias gladius*) caught, loggerhead (*Caretta caretta*) and leatherback (*Dermochelys coriacea*) sea turtle bycatch, and sea surface temperature

## CHAPTER TWO

Reducing the ecological cost of the U.S. Atlantic pelagic swordfish longline fleet: tools  
for incorporating spatial distribution into time-area closure design



## ABSTRACT

Spatially-explicit management strategies are being used with increasing frequency to reduce fisheries bycatch, but there have been few attempts to evaluate the potential costs and benefits of these conservation techniques. I used two reserve-selection approaches to assess the efficacy of reducing bycatches of leatherback sea turtles (*Dermochelys coriacea*) and blue sharks (*Prionace glauca*) in the Atlantic swordfish longline fishery. The rank optimization approach selects areas where the fleet has the highest bycatch:catch ratio as candidates for closure. Up to a 75% reduction in leatherback or blue shark bycatch could be achieved with a 20%-25% reduction in swordfish catches. These closed areas are widely dispersed, however, and it would be impractical to close fishing in so many isolated areas. In contrast, the spatial optimization approach preferentially chooses adjacent areas of high bycatch and low catches. I ran two algorithms (greedy heuristic and simulated annealing) with and without spatial constraints. The simulated annealing algorithm outperformed the greedy heuristic, meeting a greater proportion of the desired conservation targets more often. When I specified a high spatial constraint, the algorithms selected closed areas near the Grand Banks. These methods offer managers objective and simple means to maximize conservation benefits while minimizing costs to a fishery.

## INTRODUCTION

Conservation of marine species and ecosystems requires managing human activities, particularly fishing. Fisheries bycatch (the unintended catch of non-target species in fishing gear) is an important issue in fisheries management and marine conservation (Hall 1996, Murawski 1996, Crowder and Murawski 1998). It is possible to reduce bycatch without drastic restrictions of the fishery, as has been demonstrated by the reduction of the kill of pelagic dolphins in the eastern tropical Pacific purse-seine tuna fishery between 1986 and 1994 (Lennert and Hall 1996).

To successfully reduce bycatch, it is useful to be able to predict areas and times where bycatch is high (Hall 1996). Knowledge of the spatial and temporal variability of the bycatch is essential for developing strategies for reducing bycatch, because the variability in distribution and bycatch patterns affects the selection and effectiveness of management strategies. For example, if the spatial and/or temporal distributions of bycaught species are stable and localized, then mitigation strategies can depend on an element of predictability (Hall 1996), and measures such as time/area closures will be effective. If the species are widely distributed, or if the location of their distribution changes over time, then other management regimes such as gear modifications would be more appropriate. Spatially explicit analyses of species distribution and bycatch patterns as well as those of fishing effort will help determine the best management schemes to use.

Traditional fisheries management has often focused on controlling bycatch by instituting quotas, modifying gear, or reducing effort. However, the popularity of closing, or restricting, marine areas as a tool for managing and protecting marine species is increasing (*e.g.*, Allison et al. 1998, Lauck et al. 1998, Leidy and Moyle 1998, Ruckelshaus and Hays 1998, Russ and Alcala 1996, Dayton et al. 1995). For example, half of the recommendations to reduce the impacts of marine fishing on the ecosystem made by Dayton et al. (1995) involve marine protected areas and marine reserves. Marine protected areas (MPAs) often have the goal of either protecting biodiversity or of managing fisheries at an ecosystem-level (Lubchenco et al. 2003). The potential benefits of MPAs include conservation of species, protection of habitat, fisheries enhancement via recovery of overexploited stocks and export of species to adjacent areas outside the MPA, a bet-hedging technique against management or environmental uncertainty, as well as a place to conduct research (Agardy 1994, Allison et al. 1998, Lauck et al. 1998, Crowder et al. 2000). MPAs include different types of protection, depending on management goals. Marine reserves may be fully protected against all extractive and destructive activities (Lubchenco et al. 2003). Time-area closures can be viewed as a kind of spatio-temporally dynamic marine reserve, where the main goal is to reduce the extraction of a certain type of species or species complex caused by a specific fishery.

To implement spatially explicit conservation measures, it is necessary to know if we are protecting the right places. Marine ecosystems differ from terrestrial systems in terms of scale and the variability of processes, so the design of protected areas in the

ocean requires a new paradigm. Bohnsack et al. (Plan Development Team 1990) suggested marine fisheries reserves be instituted to protect adult populations of non-pelagic fishes, and recommended that at least 20% of the southeastern U.S. continental shelf should be included in such reserves. However, as Allison et al. (1998) and Crowder et al. (2000) point out, marine reserves *per se* are not sufficient for conservation. The effectiveness of an MPA depends on many factors, including the patterns of species dispersal and recruitment, source-sink processes, fisheries displacement, and spatial arrangement and location (Allison et al. 1998, Crowder et al. 2000).

The difficulty in creating protected areas lies in deciding how best to design them so that conservation is compatible with other possible management goals (Leslie et al. 2003), such as minimizing the impact on a fishery. There are many ways to select sites to include in a reserve (Prendergast et al. 1997), but there have been few applications of reserve-selection algorithms to date in marine systems (Leslie et al. 2003), and no pelagic ones (Leslie in press, [http://www.ecology.uq.edu.au/links/marine\\_applications\\_dec02.pdf](http://www.ecology.uq.edu.au/links/marine_applications_dec02.pdf)). In this chapter I illustrate two straight-forward, simple approaches, one based on a rank optimization and another based on spatial optimization, to design time-area fisheries closures.

The rank optimization approach tries to minimize the impact of closures on fishers while protecting species. If bycatch rates and catch rates are known (and assumed

static) per area, then fishing efficiency is calculated as the ratio of total bycatch to total catch. This information can be used to examine the trade-offs involved in bycatch-reduction techniques, including whether time-area closures would be an effective management tool. Hall (1996) developed this approach as an objective way to 1) reduce bycatch when fishing effort (as opposed to, for example, gear modifications) is the factor that is being controlled and 2) determine if time-area closures are an appropriate management tool to reduce fishing effort. Cramer (1996a) and Goodyear (1999) used this technique to assess the bycatch of juvenile swordfish and the possible utility of time-area closures to minimize billfish bycatch, respectively.

A fishing fleet is considered ecologically efficient when it has a low bycatch to catch ratio (i.e., it is a clean fishery). However, the fact that a fleet is ecologically efficient may not be enough to mitigate bycatch. If, in spite of being very efficient, (e.g., large numbers of target fish are being caught relative to bycaught species), the fleet is taking an unsustainable number of a species as bycatch, then other approaches are more appropriate. This is especially the case when the bycaught species have slow life histories (e.g., sea turtles) because they are particularly sensitive to incidental mortality in fisheries (Wooller et al. 1992, Dayton et al. 1995, Crouse 1999)

The ecological efficiency of the fleet per unit area can be further assessed by examining cumulative frequency distributions of bycatch/catch ratios (e.g., Hall 1996, Cramer 1996a, Goodyear 1999) for each bycaught species. The shape of the curve

indicates how much area to close to meet a certain management objective and whether, in fact, closures are appropriate management measures. For example, if bycatches are highly concentrated, the curve will have a steep slope and be convex, indicating that a large reduction in bycatch is possible with only a small reduction in target catch. These results can also serve to help assess the probability distribution of displaced effort. When bycatch reduction curves are combined with maps of bycatch to catch ratios, the spatial distribution of the fleet's ecological fishing efficiency can be assessed.

The spatial optimization approach also tries to minimize the impact of closures on fishers while protecting non-target species, but in this case, reducing the perimeter-to-area ratio of the closed area is paramount. Whether it is better to have a single large or several small (SLOSS) reserves has been widely debated (e.g., Diamond 1975).

Among the reasons one might want to design a large marine reserve is enforcement of reserve boundaries. If the reserve, or time-area closure, is comprised of many small, isolated patches, enforcement may prove to be impossible. If the goal is to reduce bycatch with little negative impact on the fishery but also make sure the areas that are closed are adjacent, and the number of potential sites (areas) that could go into the closure is large (10s –100s) then a spatially explicit reserve-design algorithm is warranted. Such algorithms include the greedy heuristic model and the simulated annealing model. *Greedy heuristic* algorithms focus on maximizing some metric of the conservation target while minimizing the number of sites. It starts with the site that provides greatest benefit and sequentially includes sites that add the most subsequent

(Csuti et al. 1997). This algorithm is computationally fast to run and straight forward, but it may not provide the optimal solution (Csuti et al. 1997, Leslie et al. 2003).

*Simulated annealing* algorithms try to find the optimal site configuration by selecting a random suite of sites and comparing that to another randomly selected suite of sites.

The one that provides the lowest total cost for the reserve system is kept, and compared to another new reserve configuration. The iteration continues until the optimal solution is found (Leslie et al. 2003). Simulated annealing algorithms allow for the explicit inclusion of space in the design of a reserve (Leslie et al. 2003).

In this chapter, I illustrate the use of these tools to help select areas and times closed to the pelagic longline swordfish fishery to reduce the bycatch of leatherback sea turtles (*Dermochelys coriacea*) and blue sharks (*Prionace glauca*).

The National Marine Fisheries Service (NMFS) is mandated by the Sustainable Fisheries Act, the Marine Mammal Protection Act and the Endangered Species Act to reduce bycatch of non-target species while promoting a healthy fishery. In 1997, NMFS began to address the issue of bycatch of non-targeted species in the pelagic longline fishery by developing the Fisheries Management Plan (FMP) for Atlantic tunas, swordfish and sharks, through Amendment One to the Atlantic billfish FMP (NMFS 1999). The FMP indicates that time-area closures should be a primary tool to reduce bycatch of all non-target species, an idea also recommended by Cramer (1996a) and Goodyear (1999).

The NMFS has already implemented several time-area closures in the Atlantic. To reduce the bycatch of juvenile swordfish and billfish, NMFS selected several time-area closures along the Atlantic and Gulf of Mexico coasts by plotting all discarded (usually undersized) swordfish on a map and drawing the most compact polygon around areas of high swordfish bycatch, and estimating the effects on the fishery (and on bycatch) given the closure (NMFS 2000c). A large area encompassing the Grand Banks has also been closed to the U.S. longlining fleet to reduce leatherback bycatch.

My objectives in this chapter are to demonstrate and evaluate simple tools that will allow managers to determine whether time area closures are really appropriate for this fishery and if so, when and where the fishery is inefficient (i.e., where a closure should be) and what size the closure should be.

## METHODS

### Data

The NMFS Atlantic Large Pelagic Logbook and the National Observer Program databases provide information on fishing effort and bycatch reported by vessel captains and documented by onboard fisheries observers, respectively. I used data from the Atlantic Pelagic Longline Logbook Program and the Atlantic Pelagic Longline Observer Program (1992 - 1999) in my analyses (Fig. 2.1). Observer coverage



averaged 3-5 % of the fishing trips between 1992 and 1999 (NMFS 2000b). To ensure that I only used pelagic sets, I eliminated all sets with less than 100 hooks (J. Cramer, pers. comm.), and also removed all sets with locational errors (i.e., sets that occurred on land). The data include number of hooks set, latitude and longitude of set, numbers of target and bycaught species, and information on sets (e.g., intended target species, gear type, light sticks, bait type). Because longline vessels are very flexible in terms of how they deploy their gear (e.g., depth of set, time of day, target species) and how much gear they use on a given set (i.e., consecutive sets in the same area and for the same target species do not necessarily have the same number of hooks), the unit of fishing effort in this chapter is the number of hooks set instead of the number of sets or trips. I did not standardize effort in any way (e.g., Hinton and Nakano 1996; Goodyear 2003) other than limiting the data I used to include only data from the swordfish fishery.

Longline gear is deployed in different ways (e.g., depth of gear in water column, time of day, hook-to-float ratios) depending on the intended target species. I selected only those sets which targeted swordfish to minimize the effects of different fishing strategies, and because the bycatch (including that of leatherbacks and blue sharks) is greater in both number and composition in swordfish-directed sets than in tuna directed sets (Hoey and Moore 1999, Johnson et al. 1999). I used the criteria used by the Observer Program to separate the sets in logbook data: a set was considered a swordfish-directed set if 50% or more of the hooks had light sticks (C. Brown, NMFS,

pers. comm. June 17, 2003) and it occurred at night (Hoey and Moore 1999). After examination of the gear set and haul times for each set per target species of the observer data, I defined nighttime sets as those in which the gear was started to be set between 2-11 pm (95.7% of observed swordfish sets) and beginning to be hauled between 4-9 am (95.6% of observed swordfish sets).

I used ArcGIS v. 8.3 (Geographic Information System; Environmental Systems Research Institute, Inc. 2002) to process all of the data. All of the resulting grids had a spatial resolution of  $1^{\circ} \times 1^{\circ}$  (~100 km x ~100 km) and are in a Lambert Equal Area projection. This map projection ensured that a grid cell in the north Atlantic covers the same area as one near the equator.

#### Interannual and seasonal spatio-temporal patterns

To assess the spatio-temporal distribution of fishing effort and the bycatch of leatherbacks and blue sharks and determine whether these distributions were consistent through time, I created GIS coverages (maps) and grids of fishing effort, and swordfish leatherback and blue shark catches per 1000 hooks for the overall dataset, for each year, and for each month pooled across years.

#### Rank Optimization

To assess what areas and times had the lowest bycatch to catch ratio (B:C) I calculated swordfish CPUE and leatherback and blue shark bycatch-per-unit-effort (BPUE) rates

per grid cell. CPUE and BPUE were calculated by dividing the number of swordfish (and turtles and sharks) by the number of hooks set in a given grid cell. Ecological fishing efficiencies per month, per year and for the datasets overall were then calculated for leatherbacks and blue sharks using the following equation: fishing efficiency per grid cell = [(bycatch rate)\* effort] per grid cell / [(target catch rate)\* effort] per grid cell (Hall 1996, Cramer 1996a). This equation is equivalent to: [number of turtles (or blue sharks) caught / number of swordfish kept] / grid cell, because the number of hooks and number of sets were the same in each grid cell. I then sorted the cells in descending order according to their bycatch to catch ratio. Grid cells were then removed in order from highest B:C values and the proportion of leatherbacks (or blue sharks) removed and the proportion of swordfish catch removed were calculated and graphed. I then mapped the areas that would have to be closed to the fishery if one wanted to reduce leatherback or blue shark bycatch by 25%, 50% and 75%.

### Spatial Optimization

I used two different reserve-siting algorithms, greedy heuristic and simulated annealing, contained in SITES v. 1.0 software (Ball 2000) with an ArcView GIS (ESRI 1999) interface to help identify grid cells that, if closed, would reduce leatherback and blue shark bycatch by 25%, 50% and 75%. In this analysis, the cost of the closed areas is considered in terms of swordfish caught. I ran each algorithm using the logbook data for the entire eight years (pooled data). I first ran each algorithm with no spatial constraints whatsoever [set the boundary length modifier (BLM) = 0] and, for an

extreme comparison, I then ran each algorithm to select the cells that were as clustered and adjacent (low perimeter to area ratio) as possible ( $BLM = 1$ ).

## RESULTS

### Annual trends in spatio-temporal distribution

A total of 45,581 and 1,838 sets targeted swordfish in the logbook and observer data sets, respectively. In this chapter, I present results for all of the data pooled over the eight-year time series. Supporting interannual and monthly data can be found at <http://seamap.env.duke.edu>.

### *Fishing Effort*

The number of hooks reported set each year ranged from 1,370,059 in 1998 to 3,534,152 in 1996 (Table 2.1). Fishing effort extended throughout most of the western Atlantic Ocean, from the Grand Banks southward along the shelf break to Florida and the waters off of the Caribbean and throughout the Gulf of Mexico (Fig. 2.2a). One notable exception occurred in 1998, when there was less fishing reported north of New Jersey. After 1995 effort expanded southward to the coast of Brazil. Concentrations of fishing effort were consistently located off the Grand Banks, along the U.S. coast from the eastern coast of Florida up to Georgia, along the eastern Gulf of Mexico, and between Cuba and Haiti (Fig. 2.2a).

The number of hooks *observed* set each year ranged from 76,730 in 1998 to 190,356 in 1995 (Table 2.1). The distribution of observed effort was similar to that reported in the logbooks (Fig. 2.2b). However, the annual distribution of observed sets varied considerably. For example, there were no observed sets north of New York in 1996 or 1998, and the number of observed sets in the Gulf of Mexico varied from year to year. The pattern of observed sets reflected the southward expansion of the fleet after 1995, as reported in the logbooks. Overall, there seemed to have been lower observer coverage in 1997-1999, no sampling south of the U.S. mainland in 1992, and little sampling in Caribbean during the entire eight year period.

### *Leatherbacks*

The number of leatherbacks *reported* caught each year ranged from 0 in both 1992 and 1999 to 294 in 1995 (Table 2.1). Leatherback bycatches were clustered primarily on the Grand Banks, with smaller clusters along the Florida east coast, in the western Gulf of Mexico, in the northern Venezuelan Basin, and offshore of the Lesser Antilles (Fig. 2.2c). There were higher bycatches in isolated areas southeast of Bermuda and east of the Lesser Antilles. The interannual distribution of leatherback bycatches (with the exception of 1992 and 1999) was relatively consistent. Areas of leatherback bycatches almost always occurred near concentrations of fishing effort. The number of leatherbacks *observed* caught each year ranged from a minimum of 2 in 1998 to a maximum of 52 in 1995 (Table 2.1). Leatherback distribution observed in the overall

data set followed the basic pattern of those reported in the logbooks (Fig. 2.2d). There were leatherbacks documented caught in both 1992 ( $n = 12$ ) and in 1999 ( $n = 38$ ).

### *Blue sharks*

The number of blue sharks reported caught each year ranged from 5,098 in 1998 to 67,691 in 1993 (Table 2.1). The distribution of blue shark bycatches occurs from the Grand Banks to Brazil (Fig. 2.2e). Blue shark bycatch occurred in two main areas: one in the northwest Atlantic and a smaller area offshore of South America. There were very few blue shark discards in the Gulf of Mexico, Caribbean, South Atlantic Bight, and offshore of Puerto Rico and the Lesser Antilles. The interannual distribution of blue sharks bycatches was consistent with the overall distribution of blue shark bycatch in both the logbook and observer data. The number of blue sharks *observed* caught each year ranged from 176 in 1996 to 2,887 in 1995 (Table 2.1). Blue shark bycatches were observed primarily off the Grand Banks in every year (Fig. 2.2f), and off the northeast coast of the U.S. from Cape Hatteras to New York in most years.

### Seasonal trends in spatio-temporal distribution

#### *Effort*

The seasonal migratory component of the fishery was reflected in the distribution of fishing effort reported per month in logbooks from 1992-1999 (Fig. 2.3). The month with the highest number of reported hooks set was January, followed by March, and February (Table 2.1). In some cases, the spatial distribution of reported fishing effort

was reflected in the observer data, but in others it was not. For example, although the distribution of observed effort reflected effort reported in logbooks for January, February, August, September and October, none of the high-effort concentrations were reflected in any of the monthly observer data sets.

### *Leatherbacks*

With the exception of the Grand Banks from June through October, leatherback bycatch was distributed in small, discontinuous patches. As with the logbooks, there were no observed large clusters of leatherback bycatch other than in the Grand Banks in the summer. In addition to discrepancies due to differences in spatial coverage, a notable difference is that there were more areas with higher ( $> 1$ ) leatherback bycatch per thousand hooks in the observer data than in the logbook data, perhaps due to underreporting of turtle bycatch in the logbooks.

### *Blue Sharks*

Blue shark bycatches were scattered from waters off the northeastern U.S. southeastward to tropical waters east of the Caribbean and South America from December through March. As with the distribution of leatherback bycatches, the pattern of observed blue shark bycatch did not always reflect that of the logbook data. Observed sets did not reflect the occurrence of reported bycatch farther offshore or in southern waters from November through July.

### Rank Optimization

Figure 4a shows the relationship between leatherback bycatches and swordfish kept for the logbook data. A 25% reduction of leatherback bycatch could be achieved at a cost of less than a 5% decrease in swordfish kept; a 50% reduction would require a decrease of swordfish kept of less than 10%; and a leatherback reduction of 75% would cost less than a 20% reduction in swordfish kept (Fig. 2.4a). This pattern was the same for all years.

With the exception of 1995, ecological fishing efficiency curves for the observer data are steeper than those generated with logbook data, indicating that leatherback bycatch documented in the observer program was highly clustered. Based on observer data, a 25% reduction of leatherback bycatch would cost less than a 10% decrease in swordfish kept (Fig. 2.4b). In all cases except for 1995, a 50% reduction would also lead to a decrease of swordfish kept of less than 10 %. A 75% reduction of leatherback bycatch would cost less than a 25% reduction in swordfish kept.

The areas that one would have to close to reduce leatherback bycatch by 25%, 50% and 75% using the logbook and observer data are shown in figures 4c and 4d, respectively. The general distribution of closed areas follows the distribution of leatherback bycatches shown in figures 2c and 2d. However, for both the logbook and the observer data sets, closure of areas that lead to a 25% reduction in leatherback bycatch are both widespread and discontinuous. As leatherback bycatch reduction increased to 50% and



75%, the closed cells became increasingly less isolated, particularly off the Grand Banks. However, the clusters were still distributed in disjunct patches because areas where no leatherback bycatches occurred were not included in the closures, even if they are located among many closed cells.

The graph in figure 5a shows the relationship between blue shark bycatches and swordfish kept for the data set overall for the logbook data. Up to a 50% reduction of blue shark bycatch would cost less than a 10% decrease in swordfish kept, and with the exception of 1993, a 75% reduction of blue shark bycatch would cost less than a 21% reduction in swordfish kept. This was also the case for the data when examined per year.

Using the observer data, a 25% reduction of blue shark bycatch would cost less than a 7% decrease in swordfish kept (Fig. 2.5b). In all cases except for 1995, a 50% reduction would also lead to a decrease of swordfish kept of less than 10 %. A 75% reduction of blue shark bycatch would cost less than a 24% reduction in swordfish kept.

The areas that one would have to close to reduce blue shark bycatch by 25%, 50% and 75% using logbook and observer data are shown in figures 5c and 5d, respectively. The general distribution of closed areas follows the distribution of blue shark distributions shown in figures 2e and 2f. For the logbook data, closure of areas that lead to a 25%

reduction in blue shark bycatch occurred mostly in the Grand Banks area. These closed grid cells were clustered more than the leatherback reduction areas, but were still patchy. As blue shark bycatch reduction increased to 50% and 75%, the closed cells became increasingly less isolated, particularly off the Grand Banks. However, there were still clusters which were distributed in disjunct patches. Patterns from the observer data were patchier, perhaps due to the smaller data set.

### *Seasonal trends*

The general patterns were the same when seasonal trends in areas of high bycatch and low catch were examined: the rank optimization approach selected areas of high bycatch and low target catch, but these areas were widely distributed. Up to a 75% reduction in leatherback or blue shark bycatch could be achieved with a 20%-25% reduction in swordfish catches (see <http://seamap.env.duke.edu> for figures).

### Spatial Optimization

Although the bycatch-reduction target (e.g., 25% reduction in bycatch) was never met in any run for leatherbacks, the annealing algorithm outperformed the greedy algorithm, meeting a greater proportion of the target (89-95%), especially when the spatial constraint was specified (Table 2.2). For blue sharks, the target was met twice in the annealing algorithm and once in the greedy, but the annealing algorithm still performed better, meeting a greater proportion of the target (99-100%), especially when the spatial constraint was specified (Table 2.3). Because the simulated annealing

algorithm outperformed the greedy heuristic algorithm, I will focus primarily on the results of the annealing model. Resulting maps for the greedy heuristic model can be found at <http://seamap.env.duke.edu>.

When no spatial constraints were specified, both algorithms selected many individual cells which had turtle or blue shark bycatch but very little swordfish catches (Fig. 2.6). As the amount of bycatch reduction increased from 25% to 50% and then 75%, the cells selected by the annealing model were adjacent to each other in some cases, for example off the Grand Banks, but often they were not (Fig. 2.6).

When the BLM was set to 1, specifying a low perimeter to area ratio in the selected cells, then even at a target reduction level of 25% the simulated annealing model selected eight adjacent cells that if closed, would reduce leatherback bycatch by almost 25%, and 12 cells that reduced blue shark bycatch by 25% (Fig. 2.7, Tables 2.2 and 2.3). For a 50% reduction in leatherback and blue shark bycatch, the simulated algorithm selected 17 and 34 cells, respectively, and for a 75% reduction in bycatch, the algorithm selected 44 and 62 cells. All of the closures recommended by the algorithm for these data sets occurred in the Grand Banks (Fig. 2.7).

## DISCUSSION

Fishing effort, swordfish catch, and leatherback and blue shark bycatch were stable in time and space. There was no obvious interannual variation in effort or catches, especially when looking at logbook data. Observer data exhibited more variation than the logbook data, most likely due to low sampling effort. Leatherback bycatch occurred for the most part in concentrated, predictable locations, such as the Grand Banks, so time-area closures could be an effective management tool to reduce this bycatch. The clustered distribution of the ecological cost (efficiency) curves also indicate that time-area closures would be appropriate. The curves also revealed that a substantial reduction of both leatherback and blue shark bycatch is possible with a relatively low reduction in target species catch, although this would be difficult to enforce due to the scattered distribution of those inefficient strata.

In examining the spatial optimization models, the simulated annealing algorithm outperformed the greedy heuristic algorithm. The proportion of the bycatch-reduction target met by the simulated annealing algorithm with a strong spatial constraint ranged from 89 - 95% for leatherbacks and from 99 -100% for blue sharks. The main advantage to this approach was that it selected efficient, enforceable areas useful for both species. Both the rank and spatial optimization approaches generally selected the region off the Grand Banks as the most important area to close.

### Spatio temporal stability

Overall, the fleet exhibited spatial stability through time. Individual cells blinked on and off across years, but this is to be expected. In general, the pattern of fishing and catch was stable, as reported by Cramer (1996c, 2001), Cramer and Adams (1998, 1999, 2000, 2001) and Goodyear (1999). Longline effort shifted away from fishing grounds in the southern Caribbean from 1990 and 1994 perhaps because this is a swordfish nursery area -- many swordfish caught in this area are undersized (Cramer 1996a). Goodyear (1999) found that individual cells turned on and off (were fished/not fished) at the  $1^{\circ} \times 1^{\circ}$  scale, but when he looked at the same data pooled in  $5^{\circ} \times 5^{\circ}$  cells, there was almost no turning on and off of cells. In the present study, individual cells do blink on and off, but this is most likely because I aggregated the data into the smallest scale used by Goodyear (1999). At this smaller scale, this blinking on and off of individual cells could be a reflection of the inherent variability of the oceanographic features the fleet uses to select areas to set their gear, such as the edge of the Gulf Stream.

### Logbook vs. observer program data

There was considerable difference in the spatial distribution of observer and logbook data. This is understandable given the target sampling level of 5% of the reported effort per year and quarter. This sampling level is based on available resources and on the estimated cost of sampling, not on the expected precision of the estimates made from these data (Johnson et al. 1999). However, this difference can have an impact on the

conclusions obtained from each data set in some cases. What one ends up protecting will vary depending on which data set one uses; thus, one must assess the caveats to both data sets to decide which to use.

Observer data are typically more reliable than logbook data in some respects, since logbooks tend to be notorious for under-reporting the numbers of bycaught species. In spite of this, Johnson et al. (1999) reported that although there were significant discrepancies between observer data and logbook data in terms of number of sea turtles caught in the whole pelagic longline fishery, both data sets showed the same pattern of high turtle catches in 1995. I used a subset of the longline data collected between 1992 and 1999, but I found the same trends in catches in the leatherback data, with peak catches in 1995. The results did not correspond as neatly for blue shark catches, with a peak catch in 1993 for logbook data and a peak in 1995 in the observer data. However, the second highest number of blue sharks reported in the observer data occurred in 1995.

#### Reserve-selection approaches

Rank optimization protects the fishery as much as possible by selecting the least efficient sites from the fishery point of view. The “purest” cells, i.e., those with the highest bycatches and lowest target catches, are selected first, so closing these areas should minimize the negative effects on the fleet in terms of reduction in catch. Spatially, however, this yields a suite of reserve-suitable sites that are largely

discontinuous and spread across large expanses of ocean, which is very likely to be difficult, if not impossible, to manage. Of course, if the number of sites is small, one can select different reserve configuration options to address various needs or concerns, such as site adjacency (spatial distribution). The cost curves generated from the rank optimization method are useful tool for visualizing the relationship between the reduction of target species and catches.

Spatial optimization may yield the suite of sites that are the easiest to manage, but are not necessarily the most efficient from the fleet's standpoint because it can add areas without bycatch to make the spatial management unit more clustered. This approach, however, is very powerful, particularly for regions with a large number of potential sites, because the user can modify many parameters to design a closure with the lowest overall costs (Leslie et al. 2003). Spatial constraints can be set anywhere along a gradient from none to extremely clustered by changing the values for the BLM. One can also start off with areas that must be included in final output, such as pre-existing reserves or closures. In addition, this reserve-selection tool can help to define sites that are "irreplaceable" by recording the number of times each site is selected during each iteration of the algorithm and seeing which sites are chosen more often than some user-defined threshold (e.g., 50% of the time; Leslie et al. 2003). The outputs of these algorithms include the amount of the target met, the cost of that particular reserve configuration, the area and number of sites included in that reserve configuration, as well as an Arc View map of the selected sites. Within the spatial optimization

algorithms, the simulated annealing outperformed the greedy heuristic in that a greater proportion of the target was met more often than in the greedy algorithm, regardless of what the BLM was (high or low).

#### Extension from single-species to multi-species

In a fishery that takes many different species as bycatch, an ideal approach would somehow balance the mortality of various species. Balancing species against each other and incorporating multiple species in this analysis simultaneously would also reduce conflicting management objectives (e.g., should one reduce sea turtle bycatch or juvenile swordfish bycatch?), but would require considerable input from various stakeholder communities. The approaches used in this chapter can be used to design the ideal (or suite of ideal) time-area closures that could reduce the bycatch of as many species possible. All of the algorithms used are extremely flexible, and would allow the user to extend this approach from a single-species one to include multiple species simultaneously. One could use the number of a given species caught, catch per unit effort, relative abundances of multiple species at once, habitat types (e.g., bathymetric ranges, or distances from seamounts), or perhaps weighted inversely by their conservation or reproductive status. The SITES program explicitly allows the user to set different criteria for weighting species against each other via setting their conservation or ecological status. Costs other than the number of swordfish caught at a given site can also be incorporated, such as the financial cost required to implement



one site over another, or combined target species (in the case of a multi-species fishery).

### Conclusion

Either of the reserve-selection approaches will provide reduction in bycatch, although with very different trade-offs. In the interest of implementation and enforcement feasibility, I recommend the use of a simulated annealing algorithm when designing a system of marine reserves or time area closures, especially if there are many potential candidate sites. This can be done for each species or species complex, then the resulting maps can be used to determine when and where the ideal closures (e.g., those which protect multiple species at once with the least cost to the fishery) should be placed. The major caveat with this approach is that there is no accounting for the effects that the redistribution of effort will have in areas that remain open to the fishery. This approach assumes that effort is not redistributed, which although is the easiest scenario to manage, is not realistic. However, the once the redistribution of the fleet is known (or estimated) this information can easily be put into either approach and used to define the best possible time-area closure.

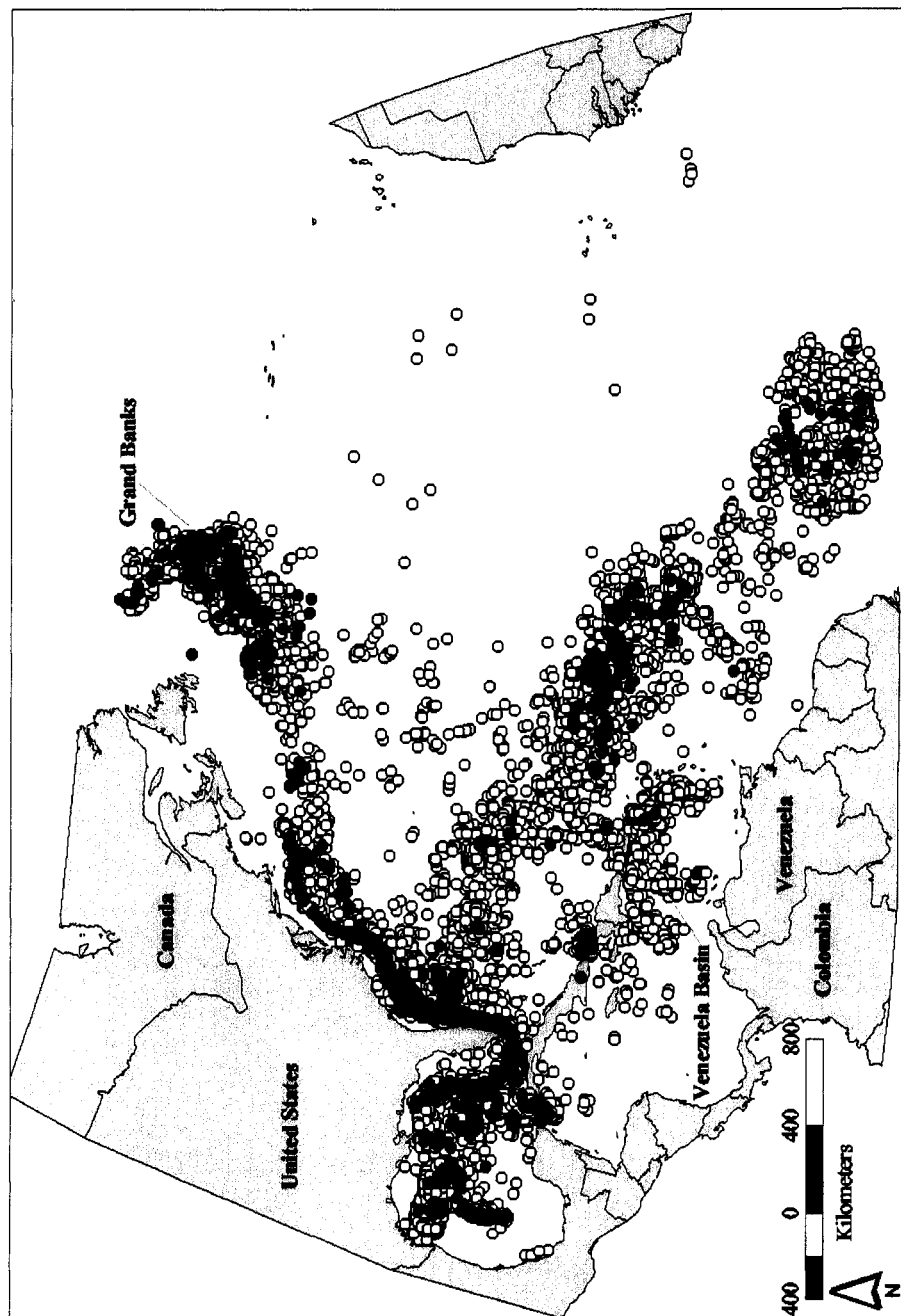


Figure 2.1. Distribution of the U.S. Atlantic pelagic swordfish longline fishery from 1992-1999. White circles represent sets reported in logbooks ( $n = 45,581$ ), black circles represent observed sets ( $n = 1838$ )

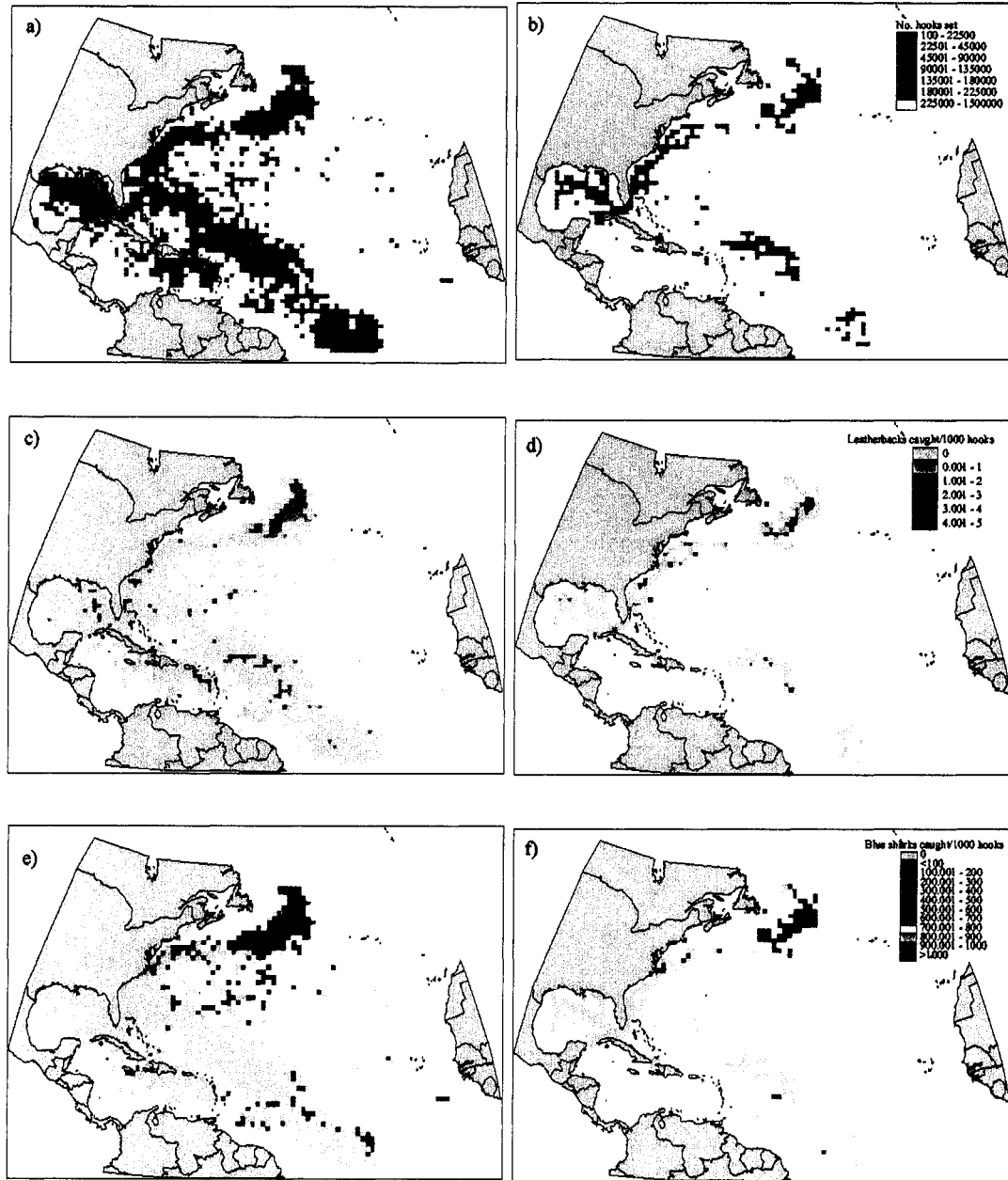


Figure 2.2. Annual distribution of fishing effort (no. hooks set), leatherback sea turtle bycatch (per 1000 hooks set), and blue shark bycatch (per 1000 hooks set) reported in logbook data and observed from 1992-1999.

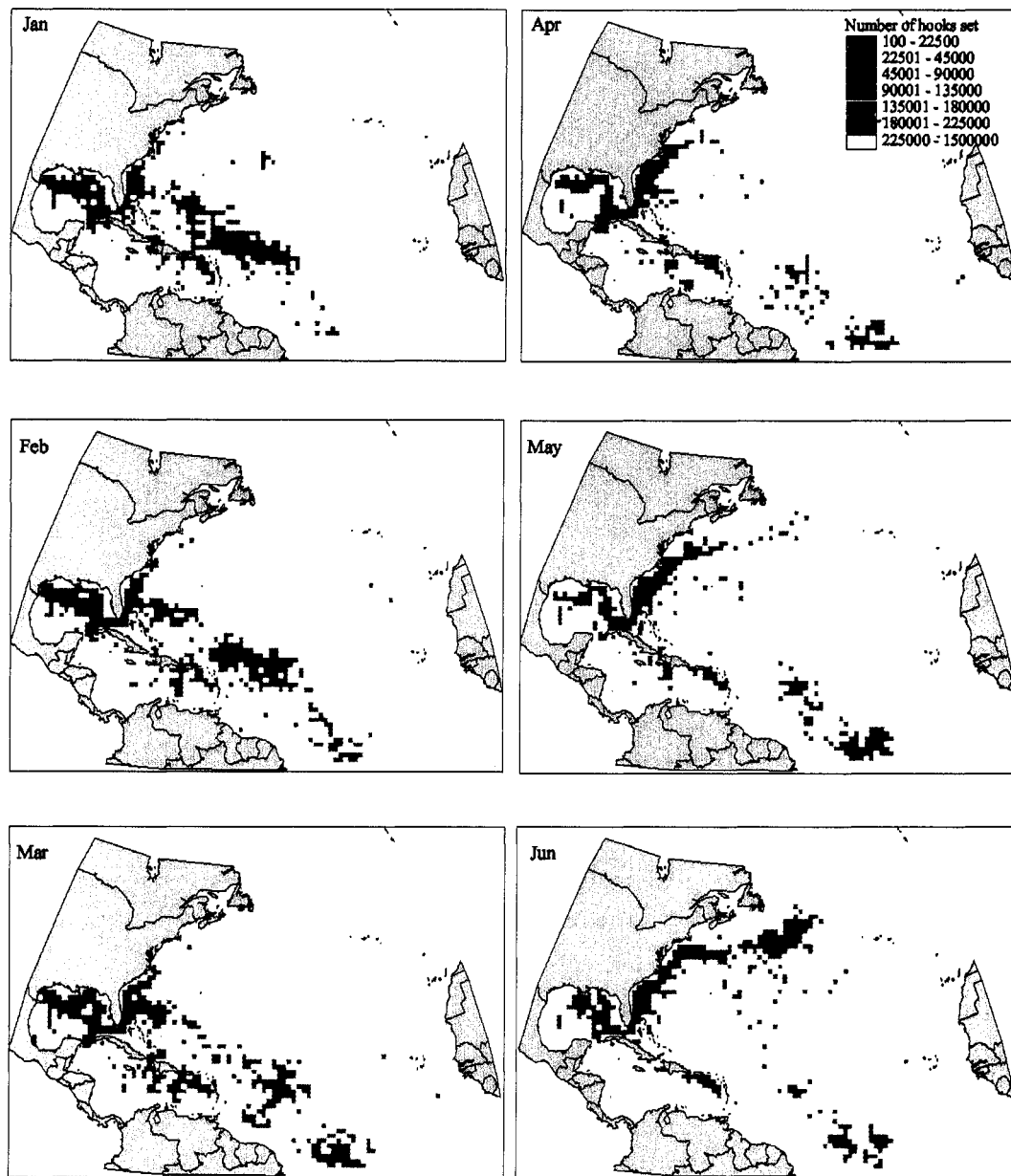


Figure 2.3. Monthly distribution of fishing effort (no. hooks set) reported in logbook data from 1992-1999.

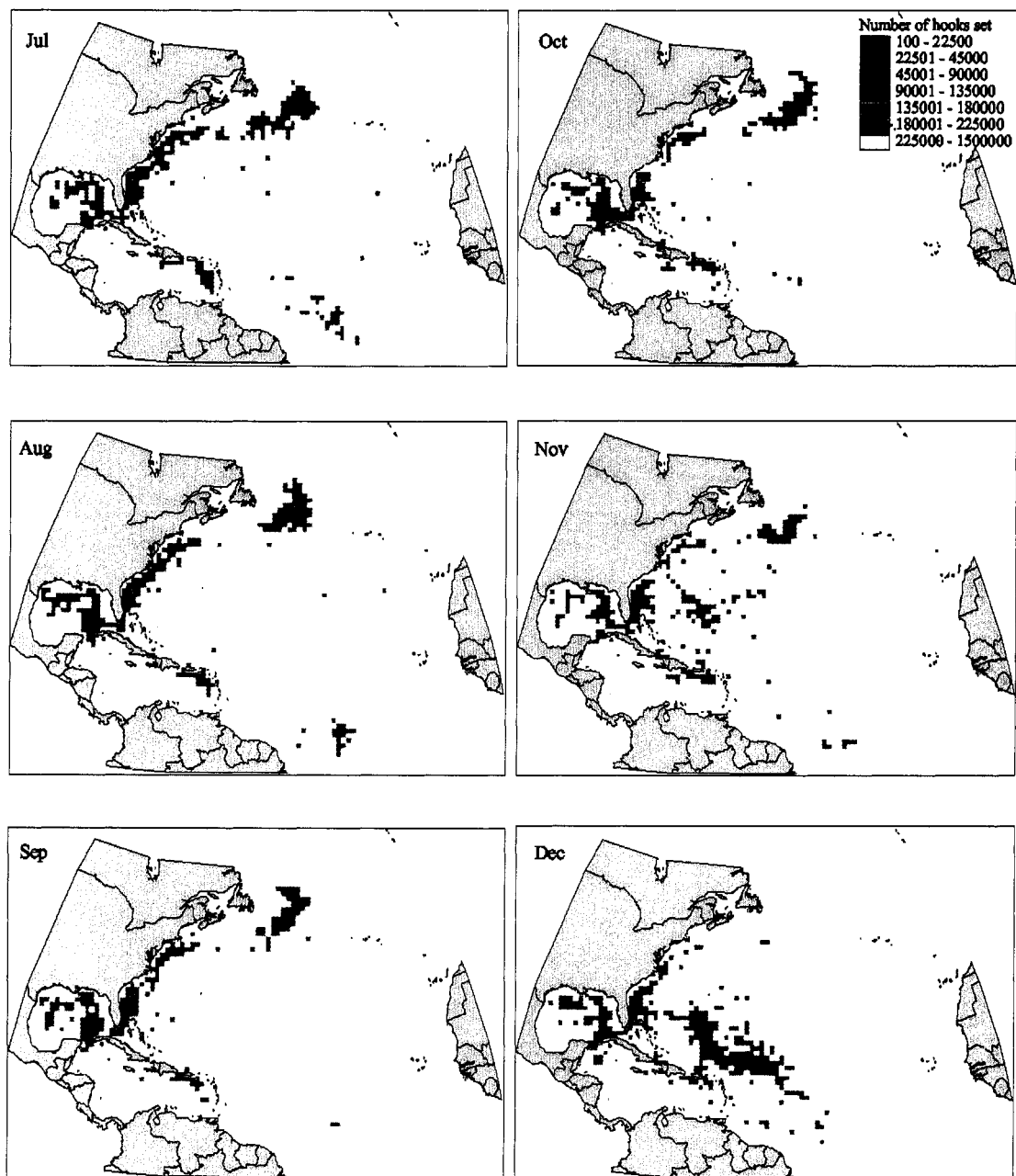


Figure 2.3. continued

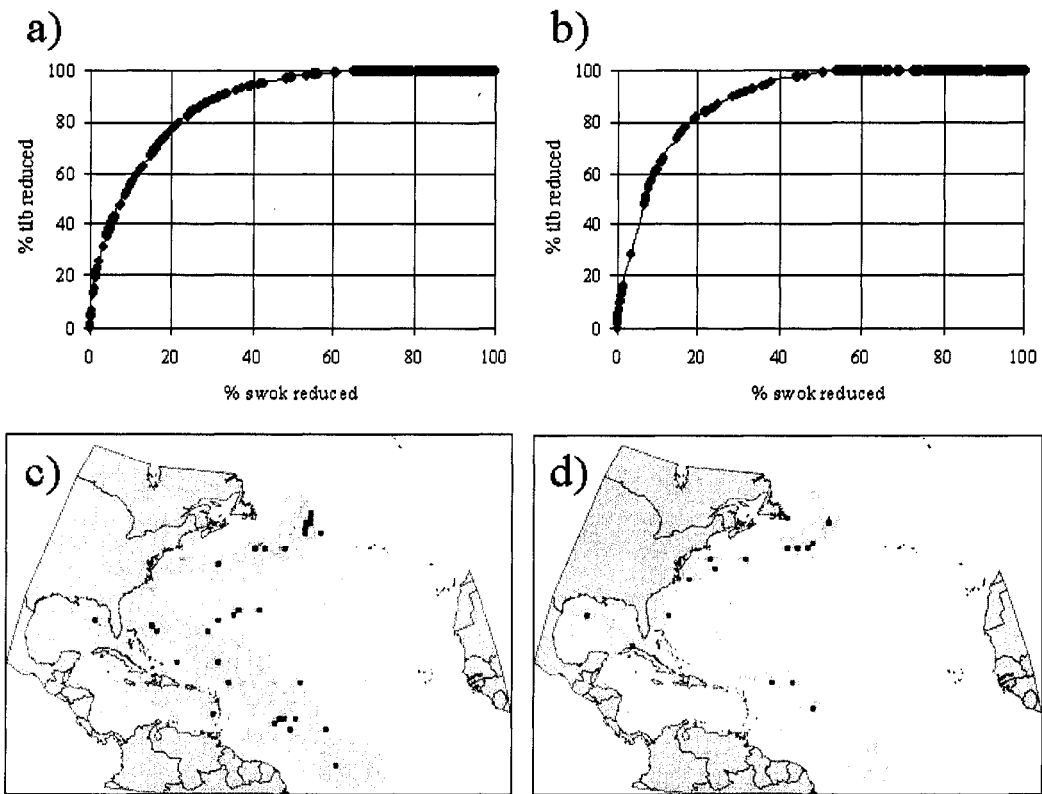


Figure 2.4. The cost per year, in terms of numbers of swordfish kept (swok), of reducing leatherback sea turtle (tlb) bycatch when ranked in descending order by bycatch to catch ratio per cell, based on a) logbook and b) observer data. Areas, ranked from least to most efficient, that would have to be closed to reduce leatherback bycatch by 25% (red), 50% (red plus orange) and 75% (red, orange plus yellow) per year based on c) logbook and d) observer data.

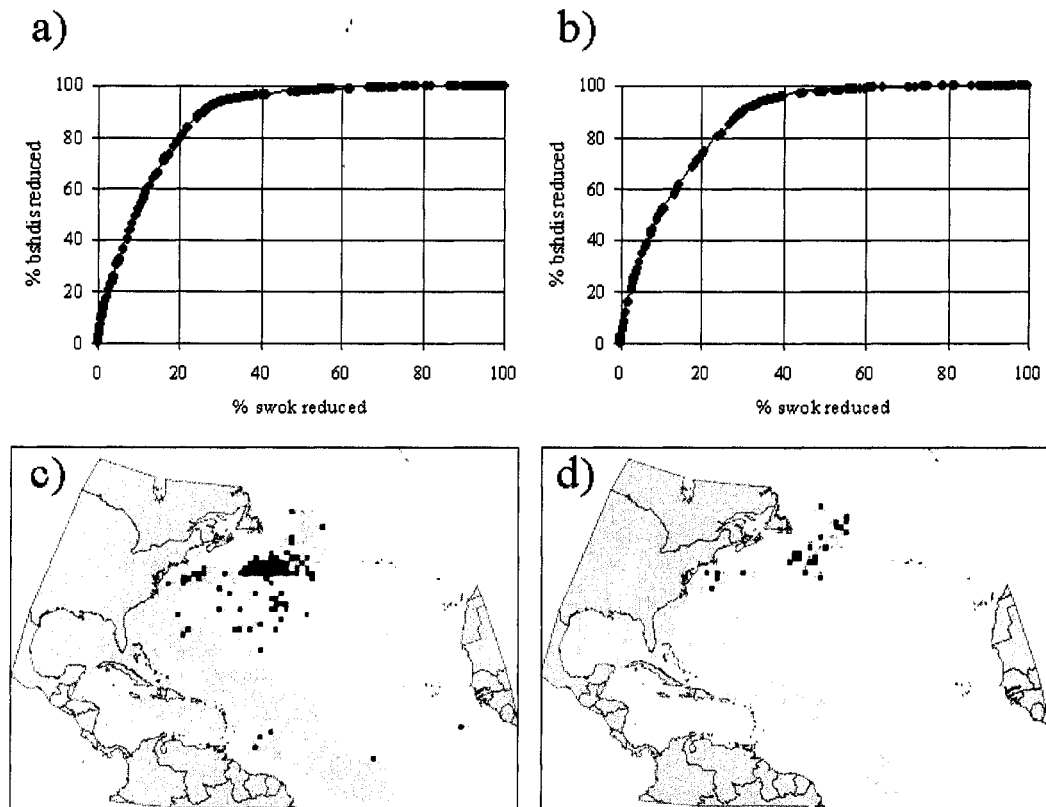


Figure 2.5. The cost per year, in terms of numbers of swordfish kept (swok), of reducing blue shark (bsh) bycatch (discards) when ranked in descending order by bycatch to catch ratio per cell, based on a) logbook and b) observer data. Areas, ranked from least to most efficient, that would have to be closed to reduce blue shark bycatch by 25% (red), 50% (red plus orange) and 75% (red, orange plus yellow) per year based on c) logbook and d) observer data

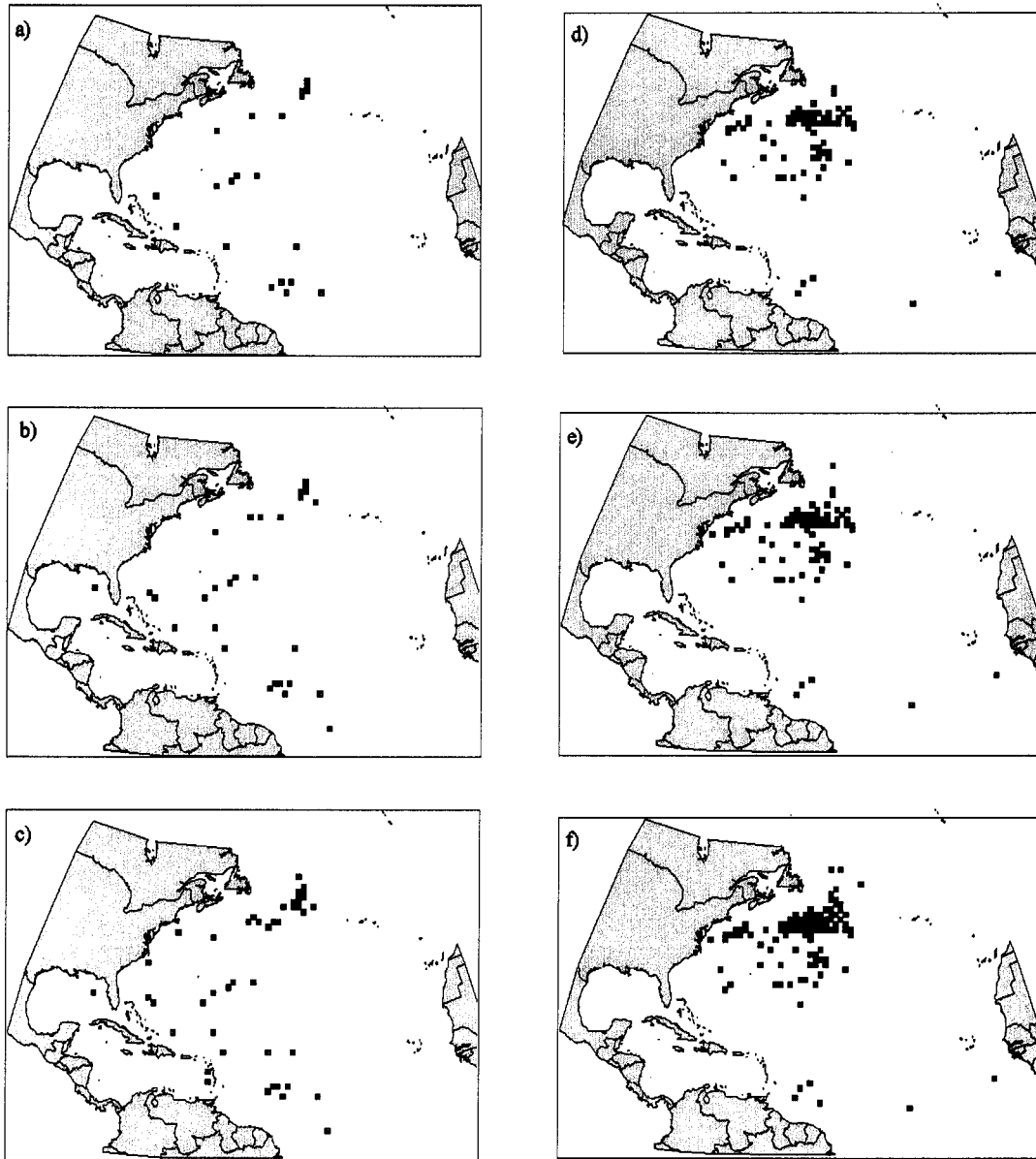


Figure 2.6. Results from simulated annealing algorithm runs with  $BLM = 0$  for a 25% (top), 50% (middle) and 75% (bottom) reduction in the bycatch of leatherback sea turtles (a-c) and blue sharks (d-f).



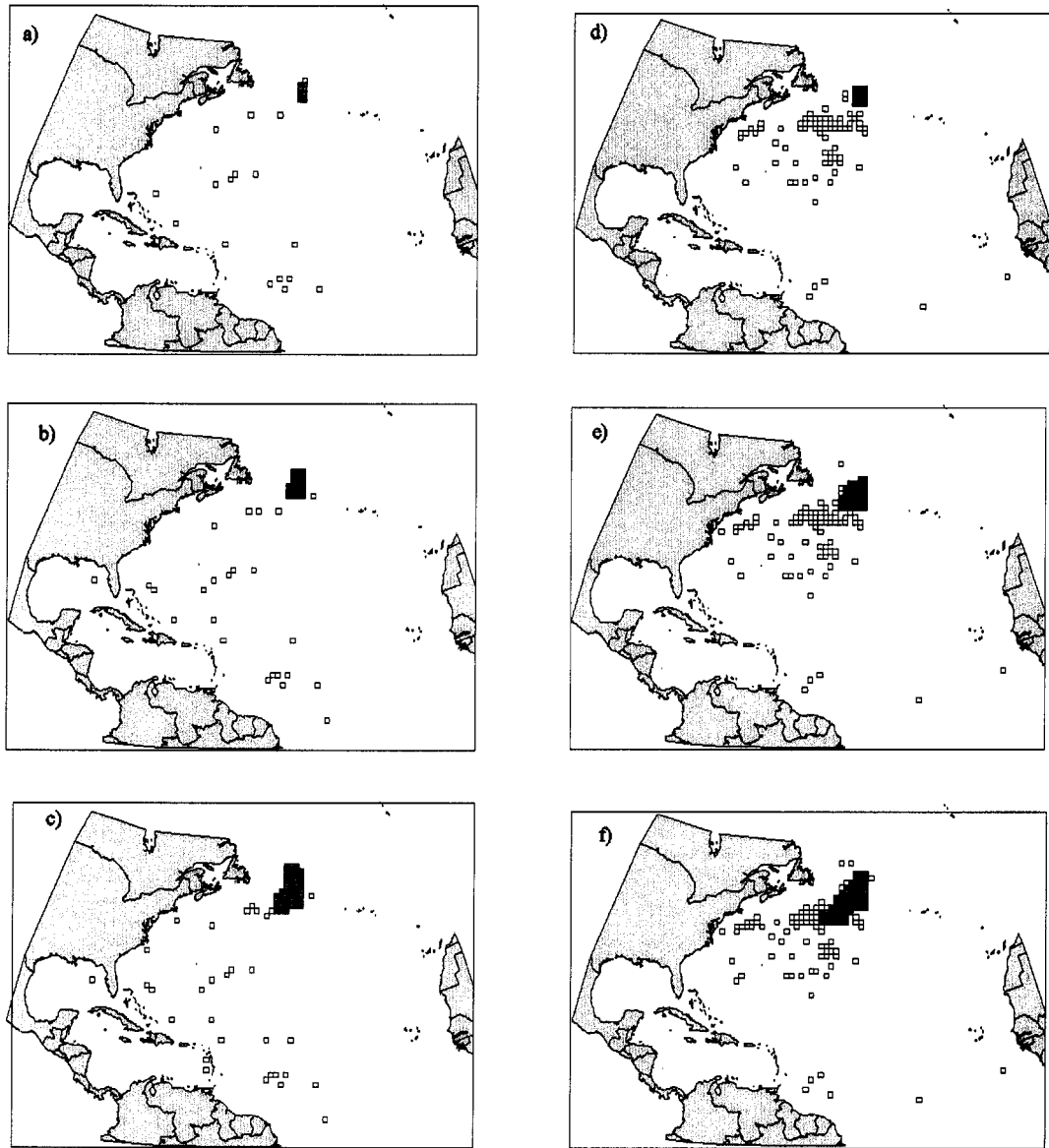


Figure 2.7. Results from simulated annealing algorithm runs with  $BLM = 1$  for a 25% (top), 50% (middle) and 75% (bottom) reduction in the bycatch of leatherback sea turtles (a-c) and blue sharks (d-f). The areas that would have to be closed given a  $BLM = 0$  are shown in white for comparison.

Table 2.1. Sample sizes used in reserve-selection analyses, derived from the NMFS observer datasets.

	No. Leatherbacks		No. Blue Sharks		No. Swordfish		No. Hooks set		No. grid cells	
	Logbook	Observer	Logbook	Observer	Logbook	Observer	Logbook	Observer	Logbook	Observer
All	651	158	319,974	9,949	365,293	26,387	20,872,015	1,007,226	1018	272
1992	0	12	57,598	738	45,549	1,830	2,677,807	82,670	413	47
1993	99	25	67,691	2,512	45,489	3,921	2,641,840	162,299	386	81
1994	128	15	60,823	713	49,182	3,413	2,857,905	120,415	375	66
1995	294	52	42,321	2,887	52,280	4,660	2,918,118	190,356	379	82
1996	50	5	38,078	176	57,189	2,623	3,534,152	124,680	463	73
1997	65	5	33,424	1,121	51,192	2,760	2,982,548	118,706	432	74
1998	15	2	5,098	385	21,895	2,571	1,370,059	76,730	301	39
1999	0	38	13,619	714	42,357	4,113	1,848,587	108,332	281	47

Table 2.1. Continued

Jan	36	11	7,300	309	38,486	2,155	2,430,539	110,035	342	56
Feb	29	2	3,325	302	30,686	2,694	2,030,061	143,696	362	69
Mar	27	3	5,043	127	35,492	2,159	2,177,477	69,552	383	41
Apr	13	1	4,900	112	23,688	1,055	1,484,747	47,123	263	28
May	19	10	5,798	277	25,168	1,689	1,659,044	75,824	278	53
Jun	29	16	53,648	930	27,656	1,953	1,838,816	80,002	332	38
Jul	142	49	48,273	3,202	35,910	3,847	1,871,464	122,054	257	54
Aug	143	32	65,708	1,593	38,115	3,255	1,834,077	112,040	229	58
Sep	134	23	50,262	973	42,088	2,139	1,768,703	69,546	203	31
Oct	55	4	50,310	1,551	31,439	3,166	1,526,796	105,619	207	56
Nov	13	1	20,041	99	15,348	1,125	900,712	30,008	227	22
Dec	11	4	3,628	50	21,048	918	1,282,725	28,269	300	20

Table 2.2. Results from spatial optimization models for leatherback sea turtles using logbook data from 1992-1999.

Algorithm	BLM	Percent Target	Target Met?	% Target Met
Annealing	0	25	No	84
Annealing	0	50	No	45
Annealing	0	75	No	56
Annealing	1	25	No	95
Annealing	1	50	No	99
Annealing	1	75	No	89
Greedy	0	25	No	84
Greedy	0	50	No	45
Greedy	0	75	No	56
Greedy	1	25	No	95
Greedy	1	50	No	98
Greedy	1	75	No	85

Table 2.3. Results from spatial optimization models for blue shark using logbook data from 1992-1999.

Algorithm	BLM	Percent Target	Target Met?	% Target Met
Annealing	0	25	No	67
Annealing	0	50	No	42
Annealing	0	75	No	58
Annealing	1	25	Yes	100
Annealing	1	50	Yes	100
Annealing	1	75	No	99
Greedy	0	25	No	67
Greedy	0	50	No	42
Greedy	0	75	No	58
Greedy	1	25	Yes	101
Greedy	1	50	No	99
Greedy	1	75	No	87

## CHAPTER THREE

Coupling GIS and foraging theory to assess the redistribution of the U.S. pelagic longline swordfish fishery

## ABSTRACT

Increasingly, managers are turning to time-area closures or other type of marine protected areas to mitigate the bycatch of non-target species in fishing activities. Often, these closures are designed and put into place without considering the redistribution of the fleet. I modeled the redistribution of fishing effort conducted by the U.S. Atlantic pelagic longline swordfish fishery given six possible time-area closures designed to reduce the bycatch of leatherback sea turtles (*Dermochelys coriacea*) and blue sharks (*Prionace glauca*) by 25%-75%. I used both logbook and observer data collected between 1992-1999. I tested whether vessels use area-restricted search patterns in deciding where to set their gear, and reallocated effort following an ideal free distribution pattern (i.e., effort was proportionally allocated to match CPUE proportions per cell). There was an inverse relationship between the distance traveled between sets and the swordfish catch per unit effort ( $R^2 = 0.3918$ ), revealing that longline fishers do exhibit area-restricted search. Bycatches of both leatherbacks and blue sharks decreased for each closure scenario given the redistribution of effort, both outside the closed areas and overall. Thus, given the model of effort redistribution, the proposed closures were effective in reducing bycatch of both leatherbacks and blue sharks.

## INTRODUCTION

The bycatch of non-target species, and how to reduce it, is a focal point in current fisheries management (e.g., Alverson et al. 1994, Hall 1996, Murawski 1996, Crowder and Murawski 1998). Fishery managers are increasingly using time-area closures, a type of marine reserve, to manage fishery resources and mitigate bycatch (e.g., Allison et al. 1998, Lauck et al. 1998, Leidy and Moyle 1998, Ruckelshaus and Hays 1998, Russ and Alcala 1996, Dayton et al. 1995) but seldom has the redistribution of the fleet been considered in conjunction with proposed closures, and the effect that this shift in effort will have on the species that are being protected.

For fisheries closures to be effective as conservation tools, managers must anticipate the redistribution of effort (Murray et al. 2000). Typically closures are implemented without a simultaneous reduction in fishing effort. To anticipate the way a fleet will relocate given a spatial closure, one must understand how fishers decide where to fish. Researchers have recently begun to address the issue of fishing behavior and how it relates to the distribution of fishing effort (Dorn 1998). Fleet spatial dynamics have been modeled from an economic point of view (e.g., Campbell and Hand 1999, Holland and Sutinen 1999, Holland 2003). However, most models have applied some version of Optimal Foraging Theory to understand different aspects of the fleet, ranging from decisions whether to retain fish of a given value (Gillis et al. 1995), when to haul and set gear (Dorn 1998), how CPUE reflects stock abundance (Hilborn and



Walters 1987, Gillis et al. 1993, Gillis and Peterman 1998), and spatial allocation of the fleet (Gillis et al. 1993, Dorn 1997).

If fishers are considered as predators, the choice of fishing location can be put into the framework of area-restricted search (ARS), and spatial allocation of the fleet can be put into the framework of the ideal free distribution (IFD, Fretwell and Lucas 1970).

Many animals exhibit ARS foraging behaviors when looking for prey in patchy environments (e.g., Ward and Saltz 1994, Sims and Quayle 1998, Veit 1999, Hill et al. 2000, 2002). An individual exhibiting ARS when searching for prey should travel in long straight lines to maximize the probability of encountering prey. When a patch is found, the individual then maximizes the chances of re-encountering that same patch as well as additional patches by traveling shorter distances and increasing its turning rate (e.g., Hildrew and Townsend 1980, Townsend and Hildrew 1980). ARS has, to my knowledge, only been applied to fishing activities by Dorn (1997).

Several researchers have applied the ideal free distribution as framework for describing the spatial allocation of fishing effort (e.g., Gillis et al. 1993, Baum et al. 2003). IFD assumes that predators distribute themselves proportionally to their prey, and that they can do so because they have perfect knowledge of the system, are free to move between patches with no constraints, and because all individuals have similar competitive abilities (Fretwell and Lucas 1970).

In this chapter, I use these aspects of foraging theory to help understand spatial fishing dynamics and model effort redistribution of the U.S. Atlantic pelagic longline fishery. I conducted this analysis using the closed areas selected by the simulated annealing algorithm in Chapter 2. The overall goal of this exercise is to reduce the bycatch of leatherback sea turtles (*Dermochelys coriacea*) and blue sharks (*Prionace glauca*). The pelagic swordfish longline fishery primarily targets swordfish (*Xiphias gladius*) and tuna (*Thunnus thynnus*, *T. albacares*, *T. obesus* and *T. alalunga*) in the U.S. Atlantic, Gulf of Mexico, and Caribbean (Hoey and Moore 1999). The U.S. fleet operates year-round and is highly mobile, ranging from 0° N, 100° W to 55° N, 10° W (Fig. 3.1). The bycatch of highly migratory species such as billfish and undersized swordfish comprises a larger portion of the total weight by species caught in longlines (Hoey and Moore 1999) and is of concern because of the potential impact on stocks of these fishes. The National Marine Fisheries Service (NMFS) has designated North Atlantic swordfish, Atlantic blue marlin (*Makaira nigricans*), Atlantic white marlin (*Tetrapturus albidus*), the large-coastal-shark complex (12 spp.), west Atlantic sailfish (*Istiophorus platypterus*), and bluefin, bigeye, and northern albacore tunas (*Thunnus thynnus*, *T. obesus* and *T. alalunga*, respectively) as overfished (NMFS 1999a). Other species of concern captured in this fishery include mahi mahi (*Coryphaena hippurus*), wahoo (*Acanthocybium solanderi*), Atlantic yellowfin tuna (*T. albacares*), and king mackerel (*Scomberomorus caballa*) (NMFS 1999a). The ecosystem-wide effects caused by the reduction of stocks of these predators are unknown. However, the removal of predators from other ecosystems appears to have substantial impacts on the

trophic structure of the communities (*e.g.*, Fogarty and Murawski 1998, Bowen 1997, Dayton et al. 1995 and references therein).

In 1997, NMFS began to address the issue of bycatch of non-targeted species in this fishery by developing the Fisheries Management Plan (FMP) for Atlantic tunas, swordfish and sharks, through Amendment One to the Atlantic billfish FMP (NMFS 1999*b*) and the development of the Atlantic Offshore Cetaceans Take Reduction Plan. In addition to complying with the requirements of the FMPs, NMFS is mandated to reduce sea turtle bycatch under the U.S. ESA.

The FMP indicates that time-area closures should be a primary tool to reduce bycatch of all non-target species. Cramer (1996) examined changes in U.S. swordfish longline landings and effort between 1990 and 1994 and identified areas with high swordfish discard rates. She found that when the fishery changed its fishing locations in the Caribbean from swordfish nursery areas to spawning areas, there was a concomitant reduction in the number of immature swordfish bycatches, but that there was no apparent shift in effort or undersized swordfish bycatches in fishing grounds near the U.S. coast with consistently high levels of bycatch. In addition, the areas that had higher bycatch/catch ratios were spatially concentrated and consistent between years (Cramer 1996). Goodyear (1999) examined the rates of billfish bycatch in relation to rates of target species catch in the U.S. longline fishery between 1990 and 1994. He also found a good deal of spatial and temporal stability in bycatch/catch ratios, and

recommended the use of time area closures to minimize the bycatch of billfishes.

Neither Cramer (1996) nor Goodyear (1999) considered the effects of the redistribution of fishing effort given a time-area closure.

In an attempt to simultaneously reduce the bycatch of billfish and small swordfish, maximize the catch of target species and not increase the bycatch of other species, NMFS (2000c) used two models to explore the effects of eight time-area closures in the Gulf of Mexico and the southeastern U.S. Atlantic. In the first model, fishers that utilized the closed areas did not fish at all, thus reducing total fishing effort bycatch and target species catch. In the second model, fishing effort was displaced randomly from closed areas to the remaining areas. Not surprisingly, under the first model the time-area closures resulted in a decrease of all bycatch. However, the model of random reallocation of effort, predicts an *increase* in sea turtle, blue marlin, white marlin and pelagic shark (including blue shark) bycatches when areas were closed to reduce the bycatch of billfish and small swordfish. NMFS predicted that, depending on where fishing effort was displaced, marine mammal bycatch would either remain constant or increase (2000c), but the agency did not address the issue quantitatively. In addition, neither model provided a realistic depiction of the effect of time-area closures. Fishing is nonrandom, with effort being directed to areas where catch rates of target species are high, unless limited by dispersal constraints, such distance from home port (Hilborn and Walters 1987, Gillis et al. 1993) or regulation. Displaced fishing effort will likely follow this pattern, with fishers moving to areas of high catch-

per-unit-effort of target species. The effectiveness of time and area closures in reducing the bycatch of protected species would be assessed most effectively using a model that represents the spatial distribution of fishing effort in a realistic manner.

I examined the effect of the time-area closures proposed in Chapter 2 on the bycatch of leatherback sea turtles and blue sharks in the Grand Banks to illustrate a simple approach in assessing how a fleet might be redistributed if a time-area closure were to be put into effect. I used the principles of ARS to understand how sets are made in this fishery, hypothesizing that if longline fishers do use ARS that subsequent gear deployments would occur closer to each other if CPUE was high, and that they would travel longer distances between sets if CPUE was low. Then I applied concepts from IFD to reallocate effort throughout the open areas nearest the closures.

## METHODS

The NMFS National Observer Program and the Atlantic Large Pelagic Logbook databases provide information on fishing effort and bycatch documented by onboard fisheries observers and reported by vessel captains, respectively. I used data from the Atlantic Pelagic Longline Observer Program and the Atlantic Pelagic Longline Logbook Program (1992 - 1999) in my analyses (Fig. 3.1). Observer coverage averaged 3-5 % of the fishing trips between 1992 and 1999 (NMFS 2000b). To ensure

that I only used pelagic sets, I eliminated all sets with less than 100 hooks (J. Cramer, pers. comm.), and also removed all sets with locational errors (i.e., sets that occurred on land). The data include number of hooks set, latitude and longitude of set, numbers of target and bycaught species, and information on sets (e.g., number of light sticks, bait type). Because longline vessels are very flexible in terms of how they deploy their gear (e.g., depth of set, time of day, target species) and how much gear they use on a given set (i.e., consecutive sets in the same area and for the same target species do not necessarily have the same number of hooks), the unit of fishing effort in this chapter is the number of hooks set instead of the number of sets or trips. I did not standardize effort in any way (e.g., Hinton and Nakano 1996; Goodyear 2003) other than limiting the data I used to include only data from the swordfish fishery.

Longline gear is deployed in different ways (e.g., depth of gear in water column, time of day, hook-to-float ratios) depending on the intended target species. I selected only those sets which targeted swordfish to minimize the effects of different fishing strategies, and because the bycatch (including that of leatherbacks and blue sharks) is greater in both number and composition in swordfish-directed sets than in tuna directed sets (Hoey and Moore 1999, Johnson et al. 1999). I used the criteria used by the Observer Program to separate the sets in logbook data (See Chapter 2).

I used ArcGIS v. 8.1 and Arc View GIS v. 3.2 (Geographic Information System; Environmental Systems Research Institute, Inc. 2001) to process all of the data. All of

the resulting maps and grids were in a Lambert Equal Area projection. This map projection ensured that a grid cell in the north Atlantic covered the same area as one near the equator. All grids had a spatial resolution of  $1^{\circ} \times 1^{\circ}$  (~100 km x ~100 km).

The Observer Program samples longline vessels in a random manner. To test the hypothesis that longline vessels fish using area-restricted search foraging techniques, I sub-selected the 10 vessels from the Observer data set that fished within a box ( $34^{\circ} 48' \text{ N}$ ,  $59^{\circ} 40' \text{ W}$  to  $55^{\circ} 42' \text{ N}$ ,  $29^{\circ} 57' \text{ W}$ ) that contained all of the possible time-area closures in Chapter 2 and which corresponds to the area southeast of the Grand Banks (Fig. 3.1). For each trip, I calculated the straight-line distance between sets using the “create poly line from points” command in the Animal Movement Extension in Arc View (Hooge and Eichenlaub, 1997) and took the average of these distances to obtain a mean distance traveled per trip. I calculated the catch per unit of effort (CPUE) as  $\text{CPUE} = \text{number of swordfish caught} / \text{number of hooks set for each set}$ , then averaged these to obtain a mean CPUE per trip. Then I conducted a negative exponential regression on the data, using the mean distance traveled as the dependent variable. I then modeled the redistribution of effort only in this area, assuming that fishers that fished off the Grand Banks would likely still fish in this area as opposed to fishing elsewhere.

I first used the six time-area closures selected by the simulated annealing algorithm in Chapter 2 and determined what proportion of the fleet (number of sets and number of

hooks) would be displaced if each of these closures were put into place, for both the logbook and the observer data. I then redistributed the number of hooks in two steps. First I made the number of hooks in each of the grid cells that corresponded to the closed areas equal to zero. Then, I added the displaced hooks (the number of hooks that fall within the boundaries of the closed areas) to the nearest open grid cells with high CPUE such that they were allocated proportionately to CPUE in a given grid cell (e.g., if 25% of the cells had the highest CPUE, then 25% of the displaced hooks were allocated evenly among those cells). I recalculated the bycatch (number of individuals per 1000 hooks) of both leatherbacks and blue sharks for each open grid cell given each of the six closure scenarios and for both data sets to assess the effect of the closures and the redistribution of effort on nearby areas. To calculate how much of the reduction in bycatch is offset by the redistribution of effort under the different closure scenarios, I added the CPUE values of all the grid cells for each of the twelve closure scenarios and compared them to the original CPUE values (i.e., no closures).

## RESULTS

The ten vessels conducted a total of 15 trips. There was an inverse trend between the mean distance traveled between sets and mean CPUE, with shorter distances traveled when more swordfish were caught [Mean distance traveled per trip =  $152984 * \exp(-22.568 * \text{mean CPUE})$ ;  $R^2 = 0.3918$ ] (Fig. 3.2). Between 3.12% - 11.70% of the total sets (5.36%-17.14% total hooks) and between 0.75%-10.28% of the total sets (1.31%-15.23% total hooks) would be displaced if the time-area closures for blue shark and



leatherbacks, respectively, that were selected in Chapter 2 were put into effect (Table 3.1).

Areas of relatively high CPUE ( $\text{CPUE} \geq 0.06$  swordfish/hooks) were immediately adjacent to the closures in each case. Recalculated CPUE for leatherbacks reached its peak ( $\text{CPUE} \leq 1$  turtle/ 1000 hooks) in the cells immediately adjacent to the closures for both data sets and all closure scenarios, with the exception of one grid cell seen at the western end of the area in the observer data for the 25% and 50% bycatch-reduction closures (Fig. 3.3). For blue sharks, CPUE also reached its peak ( $\text{CPUE} \leq 100$  sharks/1000 hooks) in the cells immediately adjacent to the closures for both data sets and all closure scenarios, with the exception of two cells with  $\text{CPUE} \leq 200$  sharks/ 1000 hooks in the 25% shark bycatch reduction closure when it was applied to the logbook data (Fig. 3.4). Overall, the bycatch of leatherbacks and blue sharks decreased for each closure scenario given the redistribution of effort. CPUE rates decreased between 28.63% and 91.66% for both species and for each closure scenario (Table 3.2). For example, under the 75% leatherback bycatch reduction scenario, leatherback CPUE is 0.854 for the entire open area -- a reduction of 91% from the original bycatch rate.

## DISCUSSION

Vessels observed in the U.S. Atlantic pelagic swordfish longline fleet appear to set their gear following area-restricted search patterns: when swordfish CPUE is low, they travel longer distances between sets than when CPUE is high (Fig. 3.2). This finding underscores the point that fishers in this fishery do not fish randomly, and validates the redistribution of effort to places of higher CPUE. Areas of high CPUE were found at the boundaries of the closures in all cases, but in spite of this, overall bycatches decreased for both species.

The inverse relationship between swordfish CPUE and distance traveled between sets suggests that fishers behave as optimal foragers, exploiting each patch such that the maximum gain is achieved before leaving it, as dictated by the Marginal Value Theorem (Charnov 1976). In a study of the mesoscale fishing patterns of factory trawlers in the Pacific Northwest, Dorn (1997) found that the decision to leave a patch was based on information from the most recent haul or two. More specifically, the decision to leave a patch was made if catches deviated from an expected catch rate, and distances between successive trawls were shorter when the vessel remained in the same patch (Dorn 1997).

The decision to reallocate effort proportionally to CPUE is supported by work done by Gillis et al. (1993), who tested whether the Hecate Strait bottom trawl fishery fished in a way that followed the IFD. They found that fishing was not random and that the spatial allocation of vessels generally followed an ideal free distribution. CPUE for

each vessel was generally equalized among vessels within a patch (Gillis et al. 1993). One of the assumptions of the IDF is that predators have perfect knowledge of the system. This assumption is more likely to be met by fishers than by non-human predators given the technology available to fishers (e.g., fish-finders, remote sensing, radio).

In this study, bycatch of both blue sharks and leatherbacks decreased with each closure scenario even after effort was redistributed (Table 3.3). The closures essentially removed areas of high bycatch, but only displaced a small percentage of fishing effort (Table 3.1). Baum et al. (2003) and Worm et al. (2003) estimated the redistribution of fishing effort in this fishery given the closure of different NMFS statistical zones -- areas that cover hundreds of thousands of square kilometers of ocean. They assumed two models: one of constant effort and one of constant catch, and redistributed fishing effort proportionally to the swordfish CPUE in each zone. They estimated that both blue shark and leatherback turtle bycatches would increase if any of the zones except the northeast distant region (NED; Fig. 3.1) were to be closed.

However, they predicted that if the entire NED were closed to protect blue sharks and leatherbacks, the bycatch of almost every other species they considered would increase because the displaced vessels would go to areas of higher diversity outside the NED (Baum et al. 2003). Displaced effort would likely be greater if the entire NED were closed completely than if just the smaller areas (i.e., the closures used in this chapter)

inside the NED were closed. Closure of the NED region, combined with the lack of nearby available fishing grounds, would indeed force all vessels to move to other areas, including biodiversity hotspots. I considered the redistribution of the longline fleet on a much smaller spatial scale, that is, smaller areas within that statistical zone. By not closing the entire NED, several areas within that larger statistical zone remain open to fishing, presumably reducing the effort that would go to biodiversity hotspots.

In this paper, I assumed fishers that fish in the Grand Banks would not go elsewhere (i.e., they would not go to another statistical zone or drop out of the fishery), but would remain in the open areas near the closures. This assumption is critical, especially given that the fleet is highly mobile. The mobility of the fleet is seasonal, presumably because CPUEs change through space and time. For example, although the migratory portion of the fleet fishes in the Caribbean during the winter, they do not fish in the Caribbean during the summer. Instead, fleet moves north to the Grand Banks. If the fleet really does fish in areas of the highest CPUE, the fishers that already fish in the area of the Grand Banks would likely continue to do so if some of this area remained open to fishing, because this is where the highest CPUE is (chapter 2). In many cases, fishing effort is displaced to areas immediately adjacent to the closed area (e.g., Bohnsack 1996, Murawski et al. submitted). It is not unrealistic to assume the displaced fishers using the area of the Grand Banks would keep fishing near the closures given that this is an area with high CPUEs at that time of year.

The approach used in this chapter is relatively simple. One advantage is that it uses data that have already been collected by the management agency, thus involves no additional expenditures for data collection. The predictions of this model can be tested simply by continuing to collect the data. If I had included data regarding landing ports (ports where vessels depart from to go fishing) I could have used the same methodology to model the redistribution of the entire fleet throughout the fishing grounds.

Reserve placement is crucial. If improperly planned, or if effort redistribution is not considered, the closure might have worse impacts on many other species (Allison et al. 1998, Crowder et al. 2000, NMFS 2000c, Baum et al. 2003). Time-area closures must be assessed on all of the species that could be affected, or else we run the risk of harming one species while saving another. The approach outlined in this chapter can and should be expanded to include as many species as affected by the longline fishery by simply recalculating the new BPUEs for that species, or for some metric of all affected species considered simultaneously (see Chapter 2 and Synthesis), once effort has been redistributed.

The use of foraging theory coupled with spatially explicit tools has proven an effective combination in assessing how the U.S. Atlantic pelagic swordfish longline fishery could respond to time-area closures in the Grand Banks.

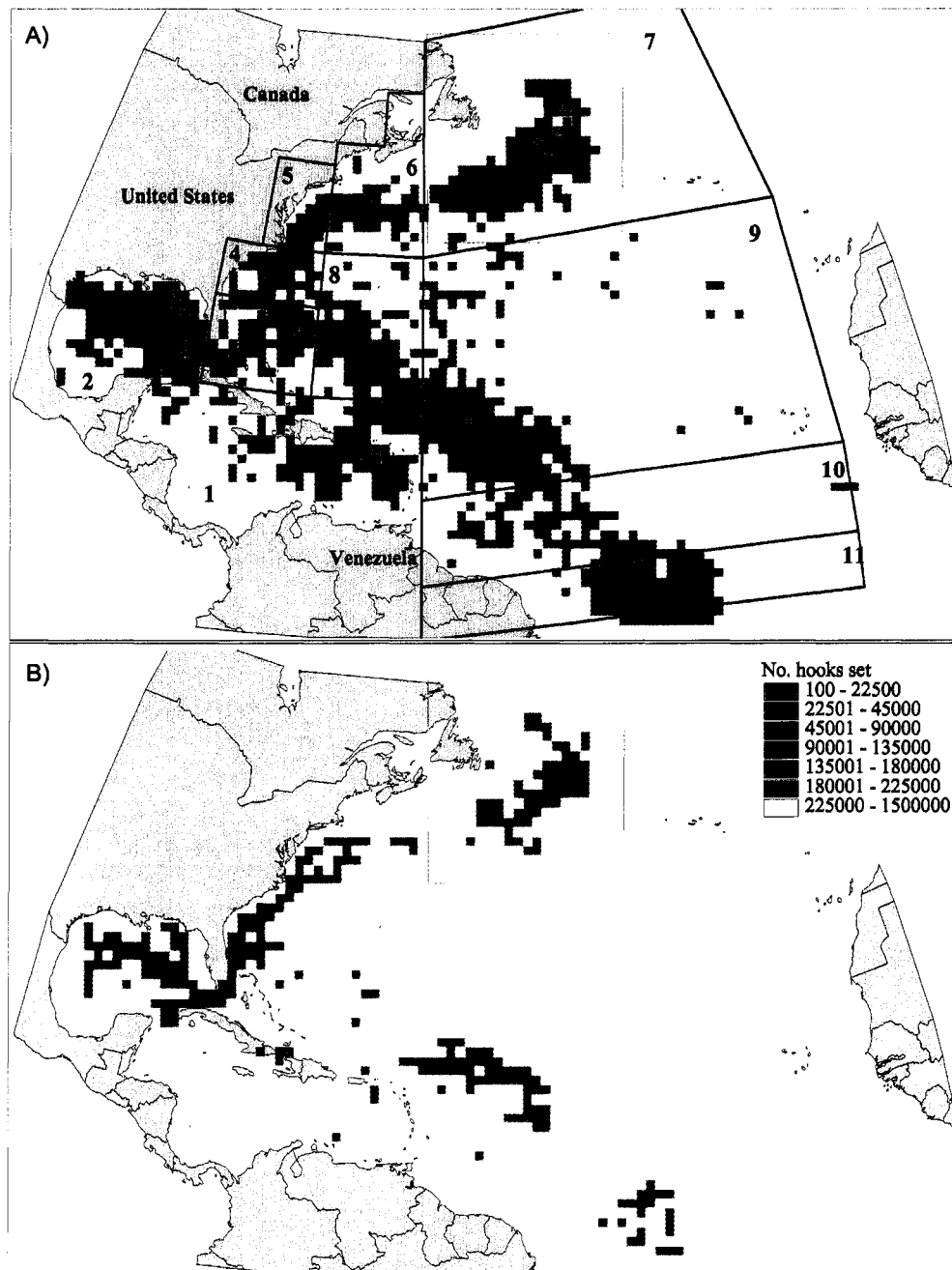


Figure 3.1. Reported (A) and observed (B) fishing effort (no. hooks set) by the U.S. Atlantic pelagic swordfish longline fishery from 1992-1999. Red box is area where fishing effort was redistributed. Black polygons represent NMFS statistical zones: 1 Caribbean; 2 Gulf of Mexico; 3 Florida East Coast; 4 South Atlantic Bight; 5 Mid Atlantic Bight; 6 Northeast Coastal; 7 Northeast Distant (NED); 8 Sargasso; 9 North Central Atlantic; 10 Tuna North; 11 Tuna South.

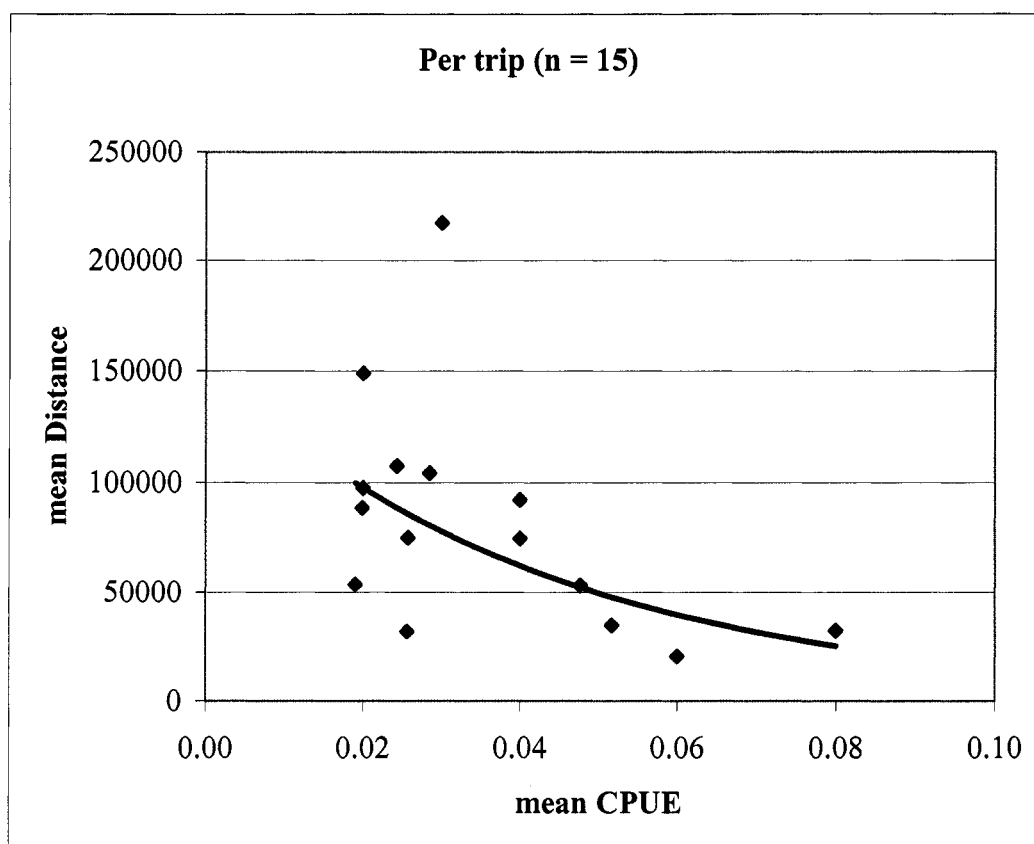


Figure 3.2. Relationship between distance traveled between sets (meters) and swordfish CPUE (No. swordfish/ No. hooks). Model: [Mean distance traveled per trip =  $152984 * \exp (-22.568 * \text{meanCPUE})$ ];  $R^2 = 0.3918$ .

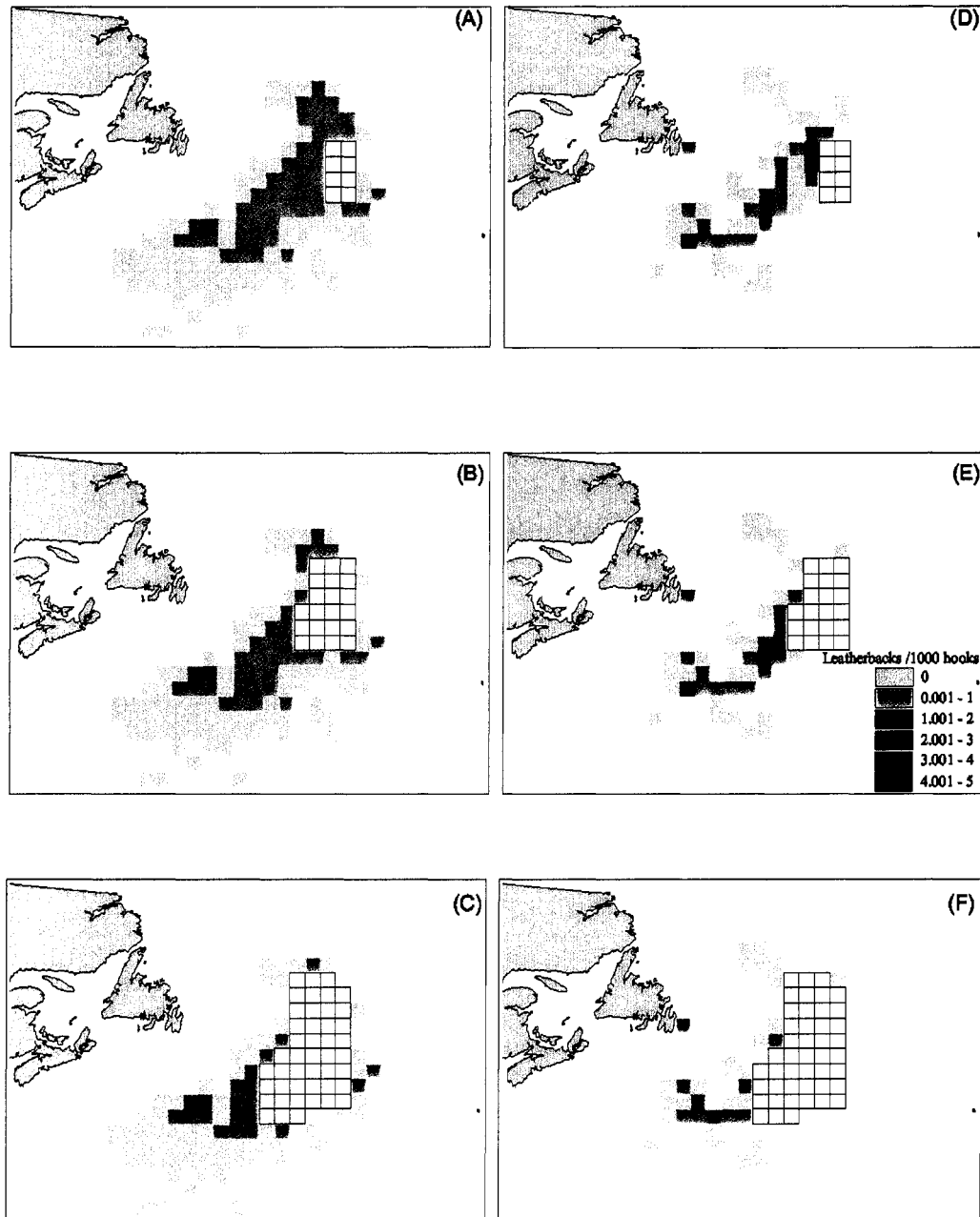


Figure 3.3. New catch-per-unit-effort for leatherback sea turtles once proposed time-area closures are in effect. A) Logbook and (D) observer data, closure to reduce bycatch by 25%; B) Logbook and (E) observer data, closure to reduce bycatch by 50%; C) Logbook and (F) observer data, closure to reduce bycatch by 75%. White squares are closures from Chapter 2.



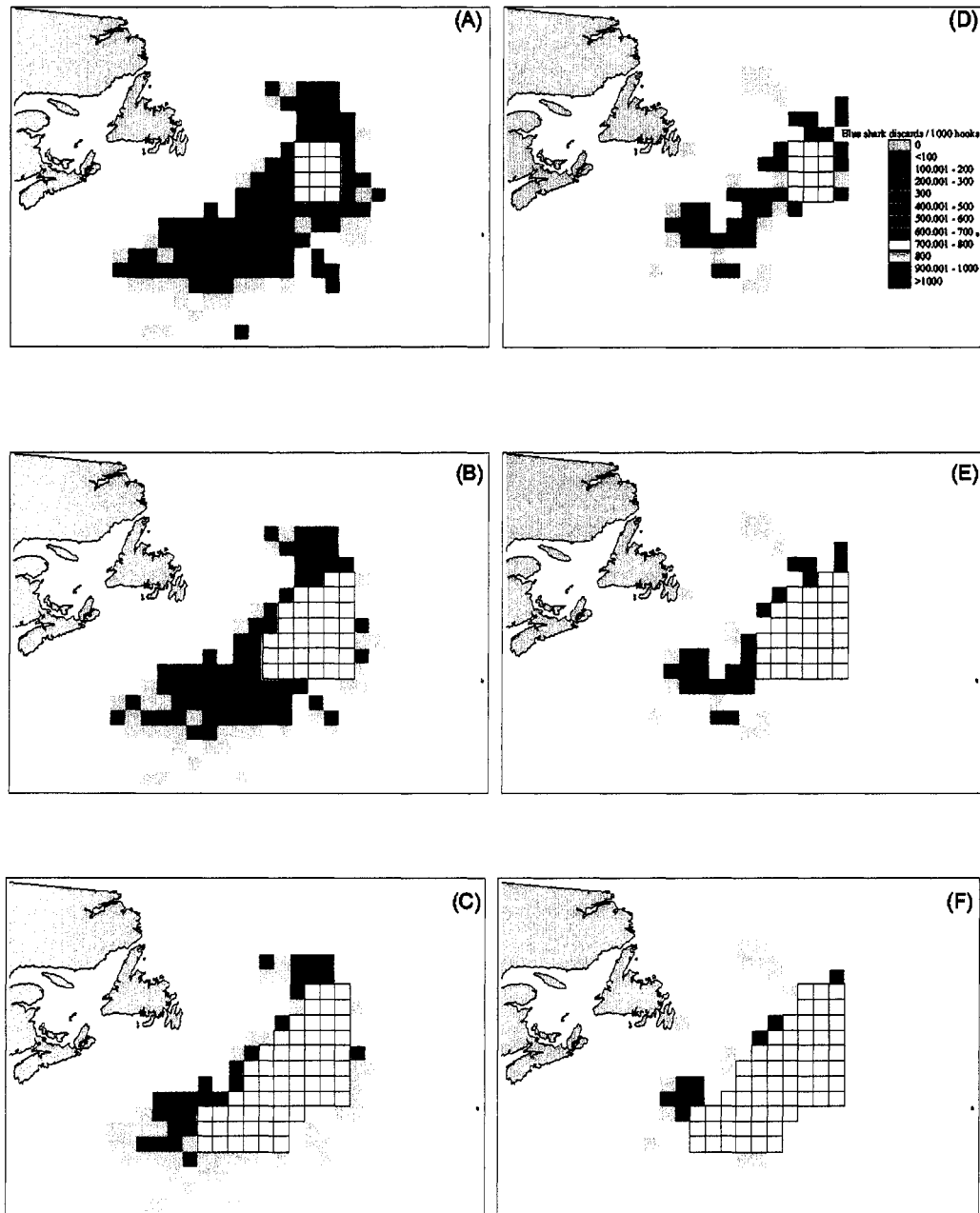


Figure 3.4. New catch-per-unit-effort for blue sharks once proposed time-area closures are in effect. A) Logbook and (D) observer data, closure to reduce bycatch by 25%; B) Logbook and (E) observer data, closure to reduce bycatch by 50%; C) Logbook and (F) observer data, closure to reduce bycatch by 75%. White squares are closures from Chapter 2.

Table 3.1. Proportion of displaced effort given closures designed by simulated annealing algorithms. Swordfish sets, 1992-1999, US Atlantic Pelagic Longline Observer Program and Logbook data set. 45,581 total swordfish-directed sets (20,872,015 hooks) in Logbook data, 1838 swordfish directed sets (1,007,226 hooks) in Observer data.

Species	% bycatch reduction	Data set	No.		No.	
			displaced sets	% total sets	displaced hooks	% total hooks
Blue shark	25	Observer	113	6.15	93,273	9.26
	50	Observer	183	9.96	148,163	14.71
	75	Observer	215	11.70	172,613	17.14
	25	Logbook	1,422	3.12	1,118,265	5.36
	50	Logbook	2,655	5.82	2,051,431	9.83
	75	Logbook	3,836	8.42	2,974,346	14.25
Leatherback	25	Observer	50	2.72	42,021	4.17
	50	Observer	111	6.04	91,638	9.10
	75	Observer	189	10.28	153,387	15.23
	25	Logbook	342	0.75	272,488	1.31
	50	Logbook	1,692	3.71	1,342,657	6.43
	75	Logbook	3,279	7.19	2,564,149	12.29

Table 3.2. Leatherback and blue shark bycatch rates (total no. individuals/1000 hooks) obtained in the study area (Fig. 3.1) under different bycatch-reduction scenarios by implementing the fishery closures from Chapter 2.

Bycatch-reduction scenario	Logbook		Observer	
	Leatherback	Blue Shark	Leatherback	Blue Shark
Original (no reduction)	9.513	9451.489	19.826	2374.657
25%	5.440	4206.204	14.150	1004.21
50%	2.580	2416.239	7.269	605.871
75%	0.854	788.1232	3.061	247.259

## SYNTHESIS

Understanding the spatial and temporal patterns of fisheries and the species they affect, the factors that give rise to these patterns and the scales at which they occur is essential to reducing bycatch and maintaining ecosystems. Bycatch reduction can be viewed as a hierarchical approach. First, if species are temporally and spatially separate, then bycatch can be reduced by spatial management measures (e.g. time-area closures). If, as in the case of the turtles in chapter 1, the bycaught species and the target species co-occur, then management becomes more complicated, with solutions falling into more traditional strategies, such as a mixture of spatial measures and operational measures, quotas, or even effort reduction.

In this dissertation, I used several spatially explicit techniques to address the conservation of blue sharks, and loggerhead and leatherback sea turtles, all of which are long-lived species whose populations are at risk. The objective of this dissertation was to apply spatially explicit techniques to address the bycatch reduction of these three species in the U.S. pelagic longline fishery and assess the potential success and drawbacks of these techniques. More so than deriving the ideal solution to the problem of bycatch in the longline fishery, I wanted to illustrate an approach to addressing the issue. With the exception of the environmental data used in chapter one, I used data that the NMFS already has in hand. All of the environmental data are available free of charge, and most of it can be obtained readily from the internet. I applied tools that have been used in other systems or for other applications, but not in the marine realm: In chapter one, I used a spatially-explicit statistical technique coupled with GIS

technology and remotely sensed data to understand the relationships between target catch, bycatch, environmental variables (i.e., habitats) and fishing effort. In chapter two, I used two different approaches, one of which was spatially explicit, coupled with GIS, to select time-area closures that have the lowest possible cost (in terms of reduction of target catch) to the fishery. In chapter three, I used GIS and foraging theory to illustrate how fishing effort could be redistributed once a time-area closure is put into place.

Chapter one illustrates the importance of including spatial information in an analysis of species' relationships with their environment. In this chapter, I examined the correlation between the numbers of bycaught leatherbacks and loggerheads, swordfish catches, several environmental variables and space in both the Atlantic and the Pacific Oceans to assess whether turtle bycatches occurred in different habitats (i.e., were spatially separate) than fishing effort and swordfish catches. I found that both loggerheads and leatherbacks in the Pacific were captured in the same environments and places as swordfish, which argues against the use of a spatial approach to bycatch reduction. Operational mitigation measures are likely to prove more useful in this case. In the Atlantic, there was some separation between turtle catches and swordfish catches by temperatures. More loggerheads were caught in cooler ( $16^{\circ}$  -  $24^{\circ}$  C) than warmer ( $>24^{\circ}$  C) water temperatures, so one potential way to reduce loggerhead bycatches is to not fish in these cooler waters. Fishers use daily AVHRR SST data to decide where to fish, so it is feasible that they could set their gear in warmer waters. With the mandated

use of vessel monitoring systems currently in place, monitoring and enforcement of this strategy is not impossible, although it would require near real-time monitoring of the vessels. However, because the effect of the redistribution of the fleet to warmer waters is unknown, and because the distribution of loggerheads and leatherbacks is also not well known, shifting all swordfish effort into warmer waters might not ameliorate the conservation status of either turtle species, given that bycatches of both species were also documented in these warmer waters.

Spatial structure in the environmental data (depth, contour index, SST and SSH) and in Atlantic swordfish catch data could have confounded the results of this chapter had space (location) *per se* not been included in the analyses. One conclusion of simply conducting an analysis on the data without including space would lead to a simple reduction in effort- a measure that will reduce bycatch, but that has drastic effects on the fishery.

Chapter two illustrates how, by including spatially explicit measures, time-area closures can be designed to minimize the negative effects on a fishery while increasing the probability that a closure will succeed in achieving its goal of reducing bycatch. In chapter two I compared two different approaches to designing time-area closures to reduce bycatch of blue sharks and leatherback turtles in the Atlantic by 25%, 50% and 75%. The first approach was based on bycatch to catch ratios, and although bycatch of both species was reduced by the target amount, these areas were spatially disjunct over

wide expanses of open ocean and would likely be difficult to implement. On the other hand, the second approach was spatially explicit and selected areas that were adjacent as well as had high bycatches and low catches. In the second approach, I used two commonly used reserve-selection algorithms (greedy heuristic and simulated annealing) and compared the efficacy of the resulting closures in meeting the targeted bycatch reductions. The simulated annealing algorithm met a greater proportion of the bycatch reduction target more often, thus it appeared to be the better model in this case.

Including information about where the areas of high bycatch and low catch were proved to be extremely useful in selecting closures that could be feasibly monitored and enforced. Both the rank-optimization and spatial optimization approaches identified areas of high bycatch to catch ratios, with the main difference being that the latter approach selected areas that were contiguous, thus easier to implement, manage and enforce. Both approaches (rank and spatial optimization) are effective, especially if there is a small number of cells that could be closed. However, if there is a large number of grid cells, the spatially-explicit reserve-selection algorithms are better, simply because the ability to permute the selection of different combinations of cells is automated.

Since fishers do not fish randomly, incorporation of spatial information in any model of redistribution is essential. For my third chapter, I coupled foraging theory with GIS technology to illustrate how one might model the redistribution of fishing effort given



a spatio-temporal closure. I tested whether longline fishers exhibit area-restricted search in terms of when and how they decide to make consecutive sets within a trip. Based on the fact that fishers seem to set their gear in places that are closer together when they are in areas of higher CPUE, I used the concept that predators will distribute themselves proportionally to the prey in a patch to reallocate fishing effort to grid cells around the closure. I then recalculated the BPUE per grid cell given the redistribution of effort. Both leatherback and blue shark bycatches decreased by closing the areas selected in chapter two and redistributing the fleet to areas adjacent to the closures.

Given the hierarchical approach to bycatch reduction mentioned above, time-area closures are appropriate for reducing the bycatch of sea turtles and blue sharks, but only if there is predictability in their spatio-temporal occurrence and the target and bycaught species do not overlap in space completely. In addition, the redistribution of effort must always be considered. Given that there are turtles and blue sharks in areas outside the Grand Banks, merely closing this area will not necessarily ameliorate the effects of fishing on these species. If a large proportion of fishing effort is displaced, turtles will be captured in redistributed effort. In the case of chapters two and three, the proportion of displaced effort was relatively small, thus the bycatches did not increase once effort was displaced. I also made one critical assumption, due in part to the availability of data: that fishers would remain in the open areas near the closed areas and not travel great distances to other fishing grounds. Access to data on the port of landing, would have allowed me to have tested the effects of this assumption.

None of the approaches that I used examined more than one species at a time, although they could all have been modified to do so. In a fishery that takes many different species as by-catch, an ideal approach would somehow balance the mortality of various species, perhaps weighted inversely by their conservation status. Such an approach was beyond the scope of my dissertation, and would require considerable input from various stakeholder communities. Mantel tests can be modified to account for multiple species by relativizing species abundance per grid cell and determining whether places (grid cells) with similar species compositions have similar environments, or even whether places with high abundances have similar environments (e.g., do places with high relative abundance have high swordfish catches, or fishing effort, or SST?). For the reserve selection algorithms, there are several ways one could deal with multiple species. One way to incorporate many species in the analyses is to conduct an analysis for each species and then overlay these in a GIS, designating a closure, and estimating the redistribution of effort. However, this would be cumbersome and very labor-intensive. Another approach would be to relativize multiple species' abundances and use this value instead of numbers of turtles or blue sharks. One could also use measures of biological diversity per cell, closing off areas of higher diversity and low target catches. Hall (1998) developed a model to illustrate the relative impacts of different fishing methods, based on their effect on the bycatch of multiple species. He essentially derived an equation which expresses how much, in terms of bycatch of other species, the bycatch of one dolphin (plus 0.1 sailfish + 0.1 manta ray) is. In the case of a fishery with many species that are marketable, economic factors such as

relative value of the species, can be used to balance species against each other. We could assign a value to, for example, a sea turtle based on ecosystem or functional value, and use this value in the analysis. Instead of making the cost of fishing in a given grid cell equal to the number (or CPUE) of swordfish, one could use the combined market value of all the marketable species, or the loss of economic gain due to handling time (e.g., disentangling large amounts of blue sharks or undersized swordfish). Once one has decided on how to weight one species against another, one can redistribute effort and then recalculate the value of bycatch (or catch) per cell and assess whether the time-area closure was effective in reducing bycatch by a given target. The problem with assigning relative values to different species is that not all stakeholders will agree on the value of species that are currently not marketable, such as sea turtles, or marine mammals.

Of course, all of these approaches can help reduce bycatch in any fishery governed by any country. However, unless bycatch-mitigation strategies are conducted simultaneously on a global scale, managing the U.S. longline fleet alone will not be enough to protect these species throughout their range. Both sea turtle species and blue sharks exhibit large-scale movements throughout ocean basins, and thus are vulnerable to longlines set not only by the U.S. fleet, but to those set by the fleets of other nations in international waters.

**Are spatial management measures appropriate for highly mobile marine vertebrates?** Space matters, and it should be considered when designing management schemes, particularly because marine habitats are very dynamic and sometimes ephemeral. The inclusion of these dynamic habitats complicates analyses because they must often be conducted repeatedly for data that represent different time-periods, but this is something that, although tedious, can be done. If we use spatially explicit information to tease apart the relative effect of our variables, then we can gain understanding of the relationships between animals and their physical and biological habitats. We can also improve the design and placement of marine reserves or protected areas, as well as anticipate the effects these will have on particular species or on the ecosystem. By selecting areas that have large areas and small perimeters, we can hopefully facilitate implementation and enforcement of these closed areas. This dissertation illustrates several approaches, tools and techniques that managers can use make better decisions and thus to insure the conservation of many marine species for future generations.

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